Responses of Bird Communities Inhabiting Boreal Plain Riparian Habitats to Forestry and Fire

A Thesis

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By

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ABSTRACT

Worldwide, riparian areas are considered among the most biologically productive and species-rich habitats on the landscape and provide important breeding areas for many bird species. In the Boreal Plain ecozone of western Canada, forests adjacent to riparian areas are generally protected from forest harvesting through the retention of treed buffer strips. Riparian buffer strips are expected to provide habitat for wildlife including many passerine bird species. Recently, non-conventional methods of riparian management have been implemented in parts of the Boreal Plain with the intent of aligning forestry more closely with natural disturbance processes. How bird communities associated with these management scenarios diverge from natural disturbances and how riparian birds interact with disturbances in the adjacent upland habitat are key questions in the conservation of boreal riparian bird communities. To answer these questions, I surveyed birds inhabiting riparian areas with adjacent naturally disturbed (burned) and harvested forest to determine how bird communities differ early (1-5 years) post-disturbance and, separately, in a before-and-after harvesting study.

Riparian species associated with burned merchantable shoreline forests and riparian areas included Common Yellowthroat (*Geothlypis trichas*) and Eastern Kingbird (*Tyrannus tyrannus*). Le Conte's Sparrow (*Ammodramus leconteii*) was associated with burned riparian habitats adjacent to non-merchantable forests (e.g., bog, fen), while Alder Flycatcher (*Empidonax alnorum*) and Wilson's Warbler (*Wilsonia pusilla*) were indicative of harvested sites with larger buffers (30 m). Riparian species richness was highest in burned non-merchantable sites. Multivariate Redundancy Analysis of post-disturbance bird communities showed greater divergence in overall (riparian and upland)

community composition than one with only riparian species. This suggests reduced sensitivity of riparian birds to disturbances in forested areas compared to upland bird communities. However, a higher natural range of variability was exhibited in riparian bird community composition in post-fire sites than in post-harvested sites. This emphasizes that forest management practices do not currently fully approximate natural disturbance for boreal riparian birds.

To assess the response of bird communities in riparian habitats to forestry, I studied bird communities one year (2004) prior to forest harvest and two years (2005 and 2006) after harvest. One of three treatments, 1) 5-35% retention (0 m buffer), 2) 35-75% retention (10 m buffer with variable retention in the next 30 m), 3) 75-100% retention (50 m buffer) and unharvested reference sites, was randomly assigned to 34 wetlands. Treatments were designed to represent buffer management strategies currently applied in the Boreal Plain. Eight of 22 species showed a significant response ($p<0.1$) to treatment, year or year*treatment effects including two riparian species, the Common Yellowthroat and Song Sparrow (*Melospiza melodia*) that increased in abundance in harvested sites. Overall pre-disturbance communities diverged ($p<0.05$) over the three-year study period as shown using Multiple-response Permutation Procedures (MRPP). However, riparian bird communities did not diverge from pre-disturbance or from reference sites providing further evidence that riparian bird communities are less impacted by forestry in the adjacent upland habitats than overall bird communities. Therefore, alternative forest harvesting methods should be explored that encompass landscape-scale management including total buffer removal to maximize conservation objectives for boreal forest bird communities while attempting to maintain natural disturbance processes.

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CHAPTER 1. GENERAL INTRODUCTION

1.1 Disturbance dynamics of the boreal forest

The Boreal Plain ecozone covers greater than 650 000 km^2 of western Canada and consists of forested areas of pure hardwood (*Populus spp.*), softwood (*Picea and Pinus spp.*) or mixedwood stands and large wetland complexes (Wiken 1986). It is a dynamic system shaped by fire, disease, insect outbreaks, wind blowdown, herbivory and flooding that create multiple seral stages and stand types important for boreal avifauna (Hobson and Bayne 2000a, b, Hobson et al. 2000, Kirk et al. 1996, Schieck and Song 2006). Such natural disturbances are considered critical to the maintenance of habitats required by wildlife (Sousa 1984, Brawn et al. 2001). The diverse array of habitats created by such natural disturbances in the Boreal Plain has led to an avian community among the richest in North America (Kirk et al. 1996) where birds make up approximately 70% of the vertebrate fauna (Smith 1993).

The introduction of anthropogenic disturbances such as large-scale forestry in recent decades represents a significant change from the structural, physical and chemical characteristics of natural disturbance regimes of the boreal ecosystem. Conventional forestry practices in boreal Canada have attempted to maintain a sustained-yield harvest over time (~70 year time scale) that would potentially result in an even-aged, single-species stand composition (Hunter 1990, Bergeron et al. 2002). Increased forest fragmentation and decreased vegetation and structural heterogeneity may also result from such practices (Hunter 1999, Lindenmayer and Franklin 2002). These departures in post-harvest environments from natural variability alter habitat availability for many bird

species in the boreal forest (Hobson and Bayne 2000a, b). Declines in the populations of some species (Kirk et al. 1997, Cumming et al. 2001) and changes in the structure and composition of boreal wildlife communities in part due to forestry (Hobson and Schieck 1999, Schieck and Hobson 2000, Schieck and Song 2006) have prompted a need for changes to conventional forestry practices that incorporate maximizing biodiversity conservation (Spence 2001, Lindenmayer and Franklin 2002, Burton et al. 2006).

1.2 The Natural Disturbance Paradigm (NDP)

The Natural Disturbance Paradigm (NDP) has been hypothesized as a way of reducing the negative effects of forest harvesting on boreal wildlife. This paradigm assumes that wildlife will more readily adapt to harvesting practices that approximate patterns of natural disturbance on the landscape (Hunter 1993, Attiwill 1994, Delong and Tanner 1996, Bergeron and Harvey 1997, Johnson et al. 1998, Bergeron et al. 1999, Niemela 1999, Lindenmayer and Franklin 2002). Most attempts at adopting forestry practices that approximate natural disturbance focus on fire as a model because of its often large-scale and significant influence on structure and pattern of forests on the landscape (Rowe and Scotter 1973, Pickett and White 1985, Hunter 1999). Modifications to conventional forestry that have been advocated include cutting to natural boundaries (e.g., rivers, wetlands, and stands), leaving live residual patches within cutblocks (with patch size, configuration and distribution similar to fire severity), cutting larger areas on a shorter temporal scale (aggregating cutblocks) and harvesting at time intervals similar to natural fire frequency (Hunter 1993, Hunter 1999, Bergeron et al. 2002, Lindenmayer and Franklin 2002).

Tests of the efficacy of the NDP using birds have primarily focused on evaluating the structure of bird communities in post-fire and post-harvest upland habitats (reviewed in Hannon and Drapeau 2006, Schieck and Song 2006). Large differences in avian community composition are typically found between early (1-5 years) post-fire and post-harvest landscapes with communities generally converging 30-60 years post-disturbance (Hobson and Schieck 1999, Imbeau et al. 1999, Schieck and Hobson 2000, Simon et al. 2002). Differences in bird communities are attributed to structural incongruity of vegetation in early post-disturbance habitats with increased congruence between communities resulting from more similar habitat characteristics at later seral stages (Hobson and Schieck 1999, Schieck and Hobson 2000). With an increased area of boreal forest being harvested through time, minimizing the differences in early post-disturbance (fire and harvest) bird community composition represents an important challenge to forest managers interested in conserving species diversity and richness similar to the natural disturbances.

1.3 Riparian areas and wetlands in the boreal forest

Riparian areas are broadly defined as ecotonal habitats that occur between aquatic (e.g. wetlands, lakes, rivers) and terrestrial environments (Gregory et al. 1991). They generally extend from the water's edge to the limit of flooding and typically exhibit high spatial and temporal variability (Gregory et al. 1991, Naiman et al. 2005). Wetlands and riparian areas represent 30-50% of the Boreal Plain ecozone (National Wetlands Working Group 1997). Although most literature refers to riparian areas as the transitional habitat adjacent to streams and rivers (e.g., Gregory et al. 1991, Naiman et al. 1993, Decamps et

al. 2004, Naiman et al. 2005), the term 'riparian' is used here to refer to the ecotone between wetlands/small lakes and upland forest (e.g., Hannon et al. 2002, Macdonald et al. 2004). The term 'shoreline forest' is used in reference to forests adjacent to water bodies and may not be considered 'riparian' forest.

Biological, physical, geochemical and hydrological processes at the riparian ecotone interact to produce among the most diverse, rich and productive habitats on many landscapes (Gregory et al. 1991, Naiman et al. 1993). Riparian areas are therefore recognized as having a disproportionately high value for maintaining biodiversity (Naiman et al. 1993, Naiman and Decamps 1997, Decamps et al. 2004, Naiman et al. 2005). Worldwide, species occurring in these areas make up 38-50% of total terrestrial species richness (Sabo et al. 2005). Estimates of terrestrial species richness do not exist for the Boreal Plain.

Due to the increased availability of water, riparian areas in the boreal forest are often characterized by an increased abundance of hydrophilic grasses, sedges (*Carex* spp.), shrubs (*Alnus* spp., *Salix* spp., *Betula glandulosa*) and trees (*Populus balsamifera, Betula papyrifera, Larix laricina, Picea mariana*) than upland forested habitats (Korol 1996, Whitaker and Montevecchi 1997, Nilsson et al. 2002). Vegetation in the riparian ecotone typically shows attributes of various wetland types including marshes (emergent), swamps (thicket, deciduous, black spruce, tamarack), bogs (graminoid, shrubby, treed) and fens (graminoid, shrubby, treed; Harris et al. 1996). Forests adjacent to lakes may also possess a distinct vegetation community consisting of greater amounts of coarse woody material, more saplings and mid-canopy trees than interior forest (Harper and Macdonald 2001, but see Macdonald et al. 2004, Macdonald et al. 2006).

Topical research on bird species' richness and diversity in riparian habitats has mostly been undertaken in riverine riparian areas across North America (see Whitaker and Montevecchi 1997, Bub et al. 2004, Hanowski et al. 2005). The few studies in boreal riparian areas have investigated birds in shoreline forests and found more species and more individual songbirds than in upland forest (Macdonald et al. 2006). However, the non-forested riparian zone likely contributes greatly to the variability of vegetation structure of the riparian ecotone (Hannon et al. 2002) and may have greater influence on avian riparian communities than shoreline forest. Although data on bird community composition in riparian and wetland habitats is sparse, approximately 43% of the species that breed in the Boreal Plain ecozone use (non-forested) riparian or wetland habitats for nesting and/or foraging and include passerine, waterfowl, shorebird and marshbird species (Appendix A3).

Many species have adapted to the specific conditions of riparian and wetland habitats. For instance, the Swamp Sparrow (*Melospiza georgiana*) nests exclusively in wetland habitats and uses its relatively long legs to forage in shallow water (Mowbray 1998). The Virginia Rail (*Rallus limicola*) possesses a laterally compressed body, long toes and flexible vertebrae that allows it to move swiftly through dense emergent vegetation (Conway 1995). Other species not specifically associated with wetland habitats including the Song Sparrow (*Melospiza melodia*), utilize these areas because they are open-natured (Arcese et al. 2002). Many species that nest or forage in the riparian interface between open water and upland forest do not necessarily require these specific habitats but also use large tracts of vegetated wetland types (e.g., bogs, fens). However, bogs and fens are currently not harvested and so were not addressed in this project.

Natural disturbances play an important role in the high spatial and temporal variation in riparian vegetation. Flooding and drawdown changes the distribution and composition of riparian and emergent vegetation (Westbrook et al. 2006). Herbivory by beaver (*Castor canadensis*) and other large mammals increases herbaceous plant richness and shrub density (Donkor 1999, Wright et al. 2002, Baker et al. 2005, Martell et al. 2006). Windthrow may decrease the number of standing trees at the riparian edge (Ruel et al. 2000, Liquori 2006, Lopez et al. 2006). Additionally, fires occur in boreal riparian areas at a rate similar to upland forest (Burgess 1997, Harper and Macdonald 2001, Macdonald et al. 2004, Macdonald et al. 2006).

Policies regarding forest harvesting adjacent to boreal riparian areas reflect a compromise between their perceived ecological, economic and social importance. The conventional approach to riparian forest management is to leave strips of trees (buffers) adjacent to the riparian zone. This is intended to maintain habitat for some species (usually large game animals and fish), allow input of organic matter into aquatic systems (France 1997, Bragg and Kershner 1999), minimize the impacts on watershed hydrology (bank stabilization, reduce overland flow of sediments; France 1997, Hickey and Doran 2004), serve as migration corridors (Machtans et al. 1996, Robichaud et al. 2002, Mosley et al. 2006) and for aesthetic purposes (e.g., visual barriers; Barten 2001, Lindenmayer and Franklin 2002, Saskatchewan Environment 2005). Riparian buffer guidelines vary across jurisdictions in the Boreal Plain ecozone where widths range from no buffer (0 m) on wetlands classified as 'beaver floods' (not fish-bearing) to >100 m on large lakes with high recreational use (Manitoba Natural Resources 1996, Saskatchewan Environment 2006a, b, Alberta Sustainable Resource Development 2006, reviewed in Lee et al. 2004).

In the absence of disturbance (fire or forestry) in riparian and shoreline forests, treed buffers "escape" succession to an earlier seral stage and therefore represent an unnatural component of the landscape.

North American studies investigating the responses of birds to riparian buffer management have occurred predominantly along rivers or lakes in the northeastern United States (Hanowski et al. 2003, 2005, 2006), in Black Spruce forests in Quebec (Darveau et al. 1995, Boulet et al. 2003), in Balsam Fir forests in Newfoundland (Whitaker and Montevecchi 1999), in mountain ecosystems (Manuwal 1986, Kinley and Newhouse 1997, Hagar 1999, Pearson and Manuwal 2001, Shirley and Smith 2005) and in the Boreal Plain of central Alberta (Lambert and Hannon 2002, Hannon et al. 2002). All of these studies focused on the ability of various-sized buffer widths in retaining upland bird communities of undisturbed or pre-harvest riparian forests. Generally, they have documented declines in smaller forested buffers of species that typically utilize contiguous interior habitats (e.g., Black-throated Green Warbler (*Dendroica virens*)) and increases in total species richness on the landscape because of the diversity of habitat types (e.g., riparian, treed buffer and cutblock). While declines of interior forest avifauna with decreased buffer width have been documented, responses of birds using the non-forested riparian habitats and shoreline forests to natural (fire) and anthropogenic (forestry) disturbances are not known. Since birds inhabiting the riparian zone represent a significant proportion of boreal avifauna, this lack of information represents a critical gap in the literature and restricts the ability of forest managers to adopt practices in shoreline forests based on natural disturbance approximation. Additionally, understanding how

riparian communities are influenced by changes in upland habitat will better able management of riparian areas on a landscape scale.

Management scenarios that consider the maintenance of natural ecosystem functions (e.g., fire behavior) have recently been applied in riparian management in the Boreal Plain ecozone of western Canada (Saskatchewan Environment 2006b). This has included a reduction in the width of required contiguous buffers along riparian zones with increased residual tree retention in the area immediately adjacent to the buffer. These modifications are an attempt to reconcile buffer management with a natural disturbance model. Responses of bird communities to these buffer guideline changes are required to determine their effectiveness in producing bird communities similar to post-fire habitats. Furthermore, bird community change in response to various buffer management strategies requires investigation to determine the impact of large, reduced or eliminated buffers on riparian bird communities.

1.4 Thesis objectives

The purpose of this thesis was to determine the responses of bird communities in non-forested riparian areas and the adjacent forests in Boreal Plain habitats of western Canada to natural (fire) and anthropogenic disturbances (various buffer management scenarios). This thesis is divided into four chapters. This first chapter is a review of natural and anthropogenic disturbance processes in the Canadian boreal forest and the response of avian communities to such disturbance. Chapter 2 investigates the responses of bird communities inhabiting riparian areas and shoreline forests to fire and forestry. The purpose of Chapter 2 was to determine whether bird communities in different-sized

buffers (1-5 years post-disturbance) adjacent to wetlands and small lakes are similar to those found in post-fire riparian sites. The discussion is centered on the potential use of the Natural Disturbance Paradigm in shoreline forests to maintain bird communities similar to post-fire habitats. Results are presented to show vegetation attributes that contribute to the divergence in bird community composition in both disturbance types. In Chapter 3, I investigate how bird communities in riparian habitats respond to three levels of harvesting relative to pre-disturbance and control sites around small wetlands one year prior to and two years after harvesting. In particular, I discuss how different environmental variables are responsible for shaping bird community composition. Finally, Chapter 4 provides a synthesis of my work and recommendations for future research.

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CHAPTER 2. MOVING RIPARIAN MANAGEMENT GUIDELINES TOWARDS A NATURAL DISTURBANCE MODEL: AN EXAMPLE USING BOREAL RIPARIAN BIRD COMMUNITIES

2.1 Abstract

Forest harvesting strategies that approximate natural disturbances have been proposed as a means of maintaining natural species' diversity and richness in the boreal forests of North America. While natural disturbances impact riparian, shoreline forests and upland areas at similar rates, forests adjacent to riparian areas are generally protected from harvesting through the retention of treed buffer strips. I examined the response of bird communities to forest management guidelines intended to approximate fires around riparian areas by comparing bird community structure in early (1-3 years) post-burned and harvested (1-4 years) boreal riparian habitats and the adjacent shoreline forest. I sampled riparian areas with adjacent: 1) burned merchantable forest (n=21), 2) burned non-merchantable forest (n=29), 3) 10 m buffer with 25% retention in the next 30 m (0-50% treed retention; n=18), and 4) 30 m buffer (50-100% retention; n=21). Indicator species of burned habitats were Common Yellowthroat (*Geothlypis trichas*), Le Conte's Sparrow (*Ammodramus leconteii*) and Eastern Kingbird (*Tyrannus tyrannus*) while Alder Flycatcher (*Empidonax alnorum*) and Wilson's Warbler (*Wilsonia pusilla*) were indicators of sites with larger buffers. Highest riparian species richness was observed in burned non-merchantable sites. Burned treatments had a greater abundance of riparian and cavity-nesting species. Multivariate Redundancy Analysis (RDA) of bird communities showed greater divergence in an overall community ordination compared to

one with only riparian species suggesting less effect of fire and forestry on riparian birds than on upland birds. Post-fire sites exhibited the highest Natural Range of Variability (NRV) in bird community composition. This emphasizes the importance of fire in the maintenance of habitats for boreal riparian birds and reveals that harvesting guidelines currently do not achieve this level of diversity. The implementation of management guidelines that better incorporate the natural range of variability of post-fire riparian areas and shoreline forests is recommended.

2.2 Introduction

Increasing anthropogenic pressures in the North American boreal forest (Cumming et al. 1994, Hobson et al. 2002, Timoney 2003) have necessitated the adoption of forest management practices that maintain natural ecosystem processes (Hunter 1993, Attiwill 1994, Delong and Tanner 1996, Spence 2001). Recent research has highlighted the impacts of such anthropogenic alterations on boreal avifauna (e.g., Welsh 1987, Kirk et al. 1997, Hobson and Bayne 2000, Cumming et al. 2001, Thompson et al. 2007). Over 200 species of birds nest in the western Canadian boreal forest (Smith 1993). Many of these show their highest abundance there (Blancher 2003) and few are considered threatened (Dunn 2002). As such, conservation practices that focus on singles-species management are neither practical nor feasible, and a coarse-filter approach (e.g., one that considers multiple species habitat requirements) to forest management is required (Armstrong et al. 2003, Burton et al. 2006).

Current hypotheses proposed to maintain wildlife populations in managed forests suggest using natural disturbance regimes to guide forest harvesting operations (Hunter

1993). This Natural Disturbance Paradigm (NDP) assumes that forest biota will be more capable of adapting to anthropogenic disturbances that retain structural attributes similar to natural disturbances, to which the species have presumably adapted (Attiwill 1994, DeLong and Tanner 1996, Bergeron 1999). Fire is considered the dominant disturbance agent in the boreal forest (Rowe and Scotter 1973, Hunter 1999) and attempts to mimic natural disturbances have primarily used wildfire as a model (DeLong 2002). Fires exhibit high variability in terms of intensity, severity and post-fire landscape structure (Cumming 2001, Wang 2002, Bergeron et al. 2004, Smucker et al. 2005), and thus, represent a challenge to forest managers attempting to approximate post-fire environments. In efforts to align forest management with ecologically-based paradigms, modifications to conventional forestry practices have included cutting larger areas (aggregating cutblocks), retention of live residual trees, harvesting along natural boundaries (e.g., stands), harvesting at intervals similar to natural fire frequency and, more recently, cutting closer to water bodies (Hunter 1993, Bergeron et al. 1999, 2002, Saskatchewan Environment 2006b).

Riparian areas are considered the most productive and species rich environments on many landscapes (Gregory et al. 1991, Naiman et al. 1993, Decamps et al. 2004, Naiman et al. 2005, Sabo et al. 2005) and in the boreal forest are used by over 40% of the bird species for nesting (Appendix A3). They are subject to frequent and recurring disturbances including flooding, herbivory, wind blowdown and fires (Naiman and Decamps 1997, Ruel et al. 2000, Andison and McCleary 2002, Dwire and Kauffman 2003, Decamps et al. 2004, Baker et al. 2005, Pettit and Naiman 2007). On the presumption that they have a disproportionately high conservation value (Gregory et al.

1991, Knopf and Sampson Samson 1994, Hunter 1999, Decamps et al. 2004), riparian areas have generally been excluded from efforts to approximate natural disturbances (e.g., harvesting shoreline forest or burning them). The most common approach in the boreal forest is to leave strips of trees (buffers) adjacent to waterbodies with the primary objective of protecting aquatic environments (Hickey and Doran 2004, Lee et al. 2004). Thus, conventional management of shoreline forests using buffer retention is not consistent with efforts to implement harvesting practices that approximate natural disturbances because such disturbances occur in shoreline forests at rates similar to upland forest (Burgess 1997, Harper and Macdonald 2001, Macdonald et al. 2004, 2006).

Most North American studies of bird responses to disturbance in riparian areas have centered on the ability of buffers of different widths in providing habitat for upland-nesting birds (e.g., Darveau et al. 1995, Kinley and Newhouse 1997, Hannon et al. 2002). Additionally, tests of the Natural Disturbance Paradigm using avifauna have focused on bird communities in upland habitats (reviewed in Hannon and Drapeau 2006, Schieck and Song 2006). These studies have shown high disparity in avian community composition between early (1-5 years) post-fire and post-harvest landscapes with convergence of community composition generally occurring 30-60 years post-disturbance (Schulte and Niemi 1998, Hobson and Schieck 1999, Simon et al. 2002). In these studies, differences in early post-disturbance bird communities were attributed to dissimilarities between post-fire and post-harvest habitat structure. No previous studies have contrasted bird communities inhabiting early post-fire and post-harvest riparian and shoreline forest habitats and so it is not clear how current riparian management guidelines depart from the Natural Disturbance Paradigm for boreal forest riparian birds.

Identifying early post-fire riparian bird community structure provides a baseline for the implementation of non-conventional methods of forest management around riparian areas. With an increased area of boreal forest being harvested through time, minimizing the differences in early post-disturbance (fire and harvest) bird communities represents an important challenge to forest managers interested in conserving species diversity, richness and community composition similar to the natural range of variation. The occurrence of multiple riparian management guidelines in a relatively confined area in the Boreal Plain allowed me to investigate which buffer management strategy produced bird communities similar to post-fire habitats.

2.3 Methods and Study Design

Study sites (4 fires and 20 cutblocks) were distributed across the Boreal Plain ecozone of western Canada (Acton et al. 1998, Strong 1992; Figure 2.1) from the House River Fire near Conklin, Alberta (55^{o}76'N 112^{o}14`W) to Candle Lake, Saskatchewan (54$^{o}$73'N 103$^{o}$ 69`W). The Boreal Plain is a mosaic of various upland forest stand ages and types including hardwoods (Trembling Aspen *Populus tremuloides,* Balsam Poplar *P. balsamifera*, White Birch *Betula papyrifera*), softwoods (White Spruce *Picea glauca*, Jack Pine *Pinus banksiana*, Balsam Fir *Abies balsamea*) and mixedwoods. Wetlands cover 30-50% of the Boreal Plain and include marshes, swamps (e.g., thicket, treed- Black Spruce *Picea mariana*), fens (e.g., graminoid, shrubby, treed- Tamarack *Larix laricina*) and bogs (e.g., graminoid, shrubby, treed- Black Spruce; National Wetlands Working Group 1997).

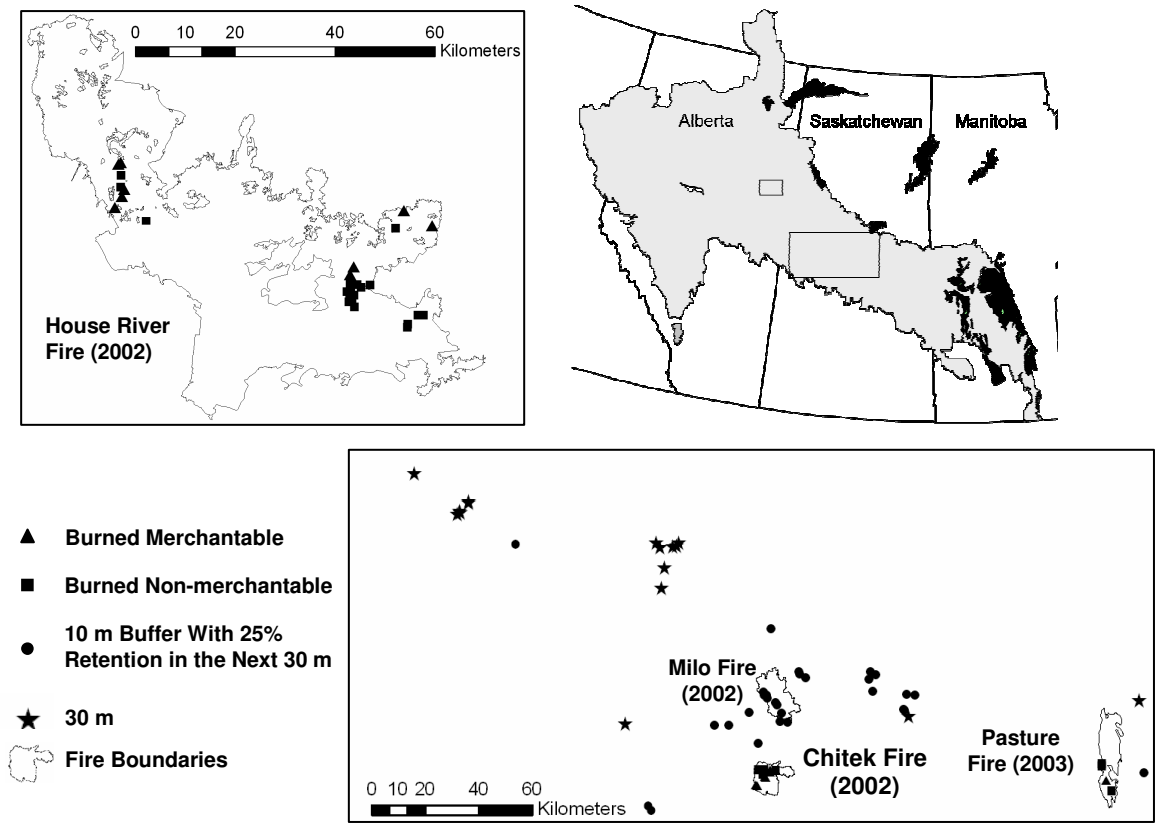

Figure 2.1. Locations of study sites within the Boreal Plain ecozone (shaded area) of western Canada. Harvested sites were interspersed between fires in Saskatchewan.

Burned merchantable sites were used to contrast different buffer widths with a natural disturbance. Burned non-merchantable sites were sampled to jointly assess, with burned merchantable sites, the natural range of variability (abundance, richness and community composition) of bird communities in early post-fire riparian habitats. Harvested sites were located in two Forest Management Areas (FMAs) in Saskatchewan with different riparian management guidelines, 10 m buffer with 25% retention in the adjacent 30 m (hereafter, 10 m buffer; Saskatchewan Environment 2006b) and 30 m buffer (Table 2.1; Saskatchewan Environment 2006a) and were interspersed between fires.

All selected wetlands were surveyed within 4 years of disturbance, had disturbance (fire or harvest) within 50 m of the high water mark, included disturbance along at least 400 m of the wetland (the length of the transect), and possessed a riparian zone (riparian vegetation) 5-15 m wide except on wetlands with non-merchantable shoreline forest where the riparian zone was up to 50 m wide. All sampled wetland were >500 m apart from each other. I attempted to survey wetlands 2-30 ha in size but included 12 larger lakes (up to 600ha) that exhibited all of the above criteria to increase the sample size.

Table 2.1 Description of treatment classes used in the study, their sample sizes (N) and mean (±2SD) wetland size (ha) from the Boreal Plain ecozone of Saskatchewan and Alberta, 2004 and 2005.

Treatment	N	Description*	Mean (±2SD) Wetland Size (ha)
Burned merchantable*	21	>70% burned merchantable forest	47.2 (±100.8)
Burned non-merchantable	29	<70% burned non-merchantable forest	11.7 (±12.8)
10 m buffer**	18	>50% merchantable forest harvested	86.4 (±193.2)
30 m buffer	21	<50% merchantable forest harvested	27.6 (±112.6)

* Merchantability for burned treatments was defined as the proportion of a 50 m digital buffer around the transect survey line considered merchantable pre-disturbance (>70% of the trees >15 cm diameter at breast height, >15 m tall, forest composition consisting of >70% hardwood or softwood species with dry soils). Percent harvested was calculated as the proportion of the merchantable timber harvested relative to pre-disturbance. Riparian area of each wetland was 5-15 m wide. Wetland sizes ranged from 2-30 ha in size with 12 larger lakes (up to 600ha).
**10 m buffer with 25% retention (see Saskatchewan Environment 2006b).

The potential influence of site-specific differences in vegetation structure and productivity on avian richness and diversity between treatments was reduced by choosing sites with similar pre-disturbance forest, and riparian composition (graminoid and shrub; Connell and Orias 1964, Waide et al. 1999). Wetlands were selected in four categories based on merchantability of shoreline forest and proportion harvested in areas where adjacent riparian and shoreline forests were disturbed by fire or harvesting (Table 2.3). The majority of sites on the landscape that met sampling criteria were surveyed and sites were therefore not selected randomly.

Merchantability of burned sites was defined as the percent of a 50 m rectangular buffer (along the survey line) created using Geographic Information Systems (GIS) that was considered harvestable (>70% of the trees >15 cm diameter at breast height, >15 m tall, forest composition consisting of >70% hardwood or softwood species with dry soils) prior to disturbance. Percent retention in harvested buffers was calculated as the proportion of the merchantable forest not harvested in a 50 m rectangular GIS buffer around the transect survey line. This allowed analysis of disturbance as both a continuous and as a categorical variable.

2.3.1 Avian Surveys

Birds were surveyed using a combined variable-width strip transect/variable-radius point count technique (Hobson and Schieck 1999). Transects 400 m in length were placed parallel to the upland-riparian ecotone to increase detections of riparian birds along the disturbed portion (burned or harvested) of the shoreline forest. Two point-counts were placed at the 50 and 350 m mark of the transect, 50 m from the water's edge to increase

the detection of upland species (Figure 2.2). Ten-minute point counts were divided into three time intervals for detection probability estimation using the count-removal method (Farnsworth et al. 2002). Distances to birds were estimated for both methods in bands of 0-25, 25-50, 50-100 or >100 m from the observer. Each site was surveyed once and point counts and transects data for each site were grouped for analyses.

Observers were trained together prior to the sampling period and were alternated between treatments to avoid confounding treatment and observer effects. After training, observers were also rated on their relative ability to identify species (low, moderate, high) for input into detection probability estimation models. Surveys started at sunrise and ceased 5 hours after sunrise during June 2004 and 2005 and were performed only on days with fair weather (no rain and winds less than 4 on a Beaufort scale). Survey start times were varied between treatments to prevent biases in species' singing rates between treatments.

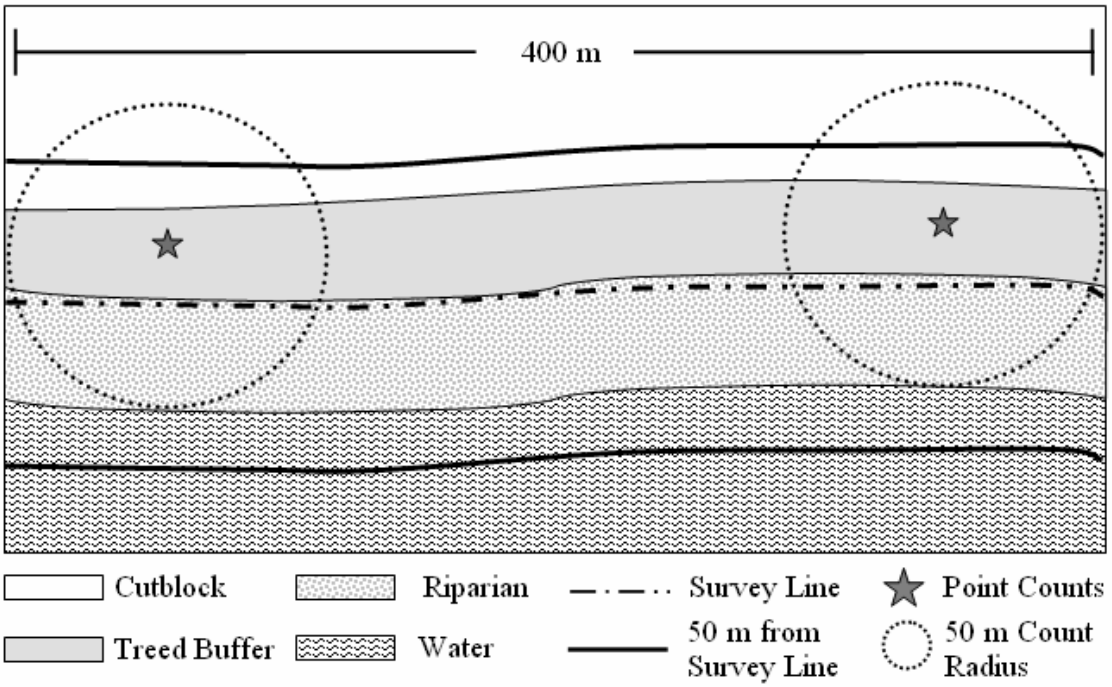

Figure 2.2. Transect (400 m) and point-count station placement in riparian areas adjacent to wetlands for bird surveys performed in the Boreal Plain of Saskatchewan and Alberta. Transects were placed at the riparian/upland forest edge and point counts were placed at 50 and 350 m along the transect, 50 m from the water's edge**.**

2.3.2 Vegetation Surveys

Generalized features of vegetation composition and structure were collected at each wetland at 100 m intervals along each survey transect (4 survey stations per transect) and encompassed an area from the open water to 50 m into the upland (Table 2.2). Vegetation sampling was conducted from mid-June to late August, 2004 and 2005. These vegetation data were averaged for the four survey stations for used in data analysis. To increase the resolution of digital forest inventories of riparian areas, vegetation features (survey line, cutblock boundary, live treed residual patches and shrub line- shrub/riparian and shrub/tree line) were mapped at each wetland using Geographic Positioning Systems (GPS) units. Additionally, digital forest inventories from three FMAs were made analogous by combining similar habitat types (polygons) into 12 categories based on species composition to reduce the number of parameters used in further data analysis (Table 2.3).

Table 2.2. Vegetation parameters measured along the riparian/upland interface at sampling areas early post-harvest and/or post-fire in the Boreal Plain ecozone of Alberta and Saskatchewan, 2004 and 2005. Data was collected at 100 m intervals along each 400 m survey transect and encompassed an area from the open water, 50 m into the upland.

Vegetation Parameter	Description
Stand type	Dominant (>70% of stand) forest type (hardwood, softwood, mixedwood, fen, bog, other)
Canopy	Dominant tree species [minimum 20% of canopy (>15 m in height)]
Canopy height	Average height of canopy
Closure	% canopy closure
Sub-canopy	5 most abundant species [minimum 20% of community (10-15m in ht.)]
Snags	% stand dead; decay stage of majority of dead trees
Shrub species	5 most abundance species (minimum 20% of community)
Shrub cover	Cover in: 1) 0-0.5 m; 2) 0.5-1.0 m; 3) 1-3 m; 4) 3-10 m height classes
Shrub height	Average height of all shrubs
Ground cover	Rank dominant cover (1-3) of herb, moss and grass
DWM	Downed Woody Material- # of downed logs >10cm diameter intercepted in 2 50 m line transects
Riparian	Average riparian width, height and dominant species
Emergent vegetation	Average emergent vegetation width, height and dominant species
Canopy burn severity	Burn severity category on features of the canopy: 0 - unburned; 1 - >60% of trees with green needles/leaves; 2 - 40-60% of trees with green needles/leaves; 3 - 5-40% of trees with green needles/leaves; 4 - <5% of trees with green needles/leaves; 5 - all trees having brown (dead) or no needles/leaves; 5 - all trees having brown (dead) or no needles/leaves; 6 - mostly broken stumps with a few standing dead trees
Ground burn severity	Burn severity category on features of the ground: 0 - no burn evidence on the forest floor; 1 - light or limited charring of duff/moss; 2 - substantially charred duff/moss; 3- extensive exposure of mineral soil
Burn height	Average height of charring on tree boles
Wetland Class	% wetland classification type of each wetland Swamp (shrubby, hardwood, softwood, mixedwood), marsh, bog, (shrubby, treed), fen (graminoid, shrubby, treed), etc.

Table 2.3. Criteria used to define habitat associations from digital forest inventories for bird communities around small wetlands with adjacent burned merchantable forest, burned non-merchantable forest, 10 m buffer with 25% tree retention in the next 30 m and 30 m treed buffer in the Boreal Plain ecozone of Saskatchewan and Alberta, 2004 and 2005. Data were entered into the analysis as proportion of each forest type of a 50 m rectangular digital buffer around the transect survey line.

Habitat Type	**Criteria**	**Species***
Hardwood	>70% hardwood	tA, bP, wB
Hardwood/Softwood	50-70% hardwood/30-50% softwood	tA, bP, wB>wS, bF
Softwood/Hardwood	50-70% softwood/30-50% hardwood	wS, bF, tA, bP>wB
Softwood	>70% Softwood	wS, bF
Other Hardwood/ Softwood	50-70% Hardwood with low merchantable softwood	tA, bP, wB, bS, tL, jP
Other Softwood/ Hardwood	50-70% low merchantable softwood	bS, tL, jP>tA, bP, wB
Other Softwood	>70% low merchantable softwood	bS, tL, jP
Bog	Softwood with wet to very wet soils	bS>tL, jP
Fen	Softwood with wet to very wet soils	tL>bS, jP
Shrub	Shrub dominated	N/A
Graminoid	Meadow/grass dominated	N/A
Water	Permanent water- wetlands, lakes, rivers	N/A

*tA- Trembling Aspen, bP- Balsam Poplar, wB- White Birch, wS- White Spruce, bF- Balsam Fir, bS- Black Spruce, tL- Tamarack Larch, jP- Jack Pine. Hardwood species <15m tall were considered shrubs.

2.4 Data Analysis

2.4.1 Pre- and Post-Disturbance Vegetation

Principle Components Analysis (PCA) was used to explore the inter-correlation between vegetation variables and to determine whether the number of variables could be reduced to fewer explanatory components. Separate PCA analyses were run for pre- and post-disturbance GIS data, field observations, and wetland classification data. Analysis of digital vegetation inventories was restricted to parameters extracted from a 50 m wide rectangular, flat-ended GIS buffer around the survey line. Pre- and post-disturbance proportion data were arcsine transformed prior to analysis to improve non-normally distributed data. Habitat data from field observations were adjusted to zero mean and unit variance $[(x - \bar{x})/SD]$ to allow comparison of variables measured in different units. The retained Principle Components were selected using the broken-stick method (McCune and Mefford 1999). All multivariate data were analyzed with PC-ORD v.4.0 (McCune and Mefford 1999) except where noted. PCA scores were compared for between-treatment differences using a simple one-way Analysis of Variance (ANOVA) in SPSS 14.0.0. A level of $p<0.05$ was used to indicate significant differences in all univariate tests except where noted.

2.4.2. Avian Analysis

2.4.2.1 Detection Probability Estimation

Farnsworth et al's. (2002) removal method was used to estimate detection probabilities for species (>80 observations) or groups of species (<80 detections of individual species; Alldredge et al. 2007) detected on point counts. Species were grouped

by song characteristics, singing frequency, behavioral traits and habitat preferences (Alldredge et al. 2007; Appendix A2). I grouped data from two study areas that included point-count data from Alberta, Saskatchewan and Manitoba collected using the same protocol in similar habitats (see 2.3.1 Avian Surveys). Detectability models were run using Huggins' (1989) closed-capture models in program R (R Core Development Team 2007) with RMark v.1.6.4 (Laake 2007), an interface to program MARK (White 2006). To account for a non-linear relationship of effort with the number of new detections on a point count, all models were fit with a natural log function of cumulative time interval (3, 5 and 10 minutes). Temporal (sampling date, time of day), observer ability, habitat, distance from observer variables (midpoint of distance band intervals) and various interaction terms (Appendix A1) were incorporated as covariates into the models to increase the precision of the estimates. Twenty-eight models were evaluated based on AICc values (Akaike's Information Criteria adjusted for small sample size; Akaike 1973) and were selected if ΔAICc was less than 4.0 (Burnham and Anderson 2002). Beta parameters were model averaged using model weights of all top models to determine the detection probability for each 10-minute point count. Detection probabilities were not estimated for species with few detections (e.g., Pine Siskin *Carduelis pinus*) and were assumed to be 100%.

Detection probabilities for birds encountered only on transects were not estimated because the layout of the transects (i.e., parallel to the riparian habitat) violated assumptions of DISTANCE sampling (Buckland et al. 2001). Therefore, further analyses using birds were performed on point counts adjusted for detection probability and unadjusted transect counts, combined. Multivariate analyses can accurately represent data

using only presence-absence measures (McCune and Grace 2002) and use of unadjusted counts should not affect the results. In addition, the nature of the riparian habitat (e.g., open) resulted in assumed high species detection rates compared to other habitat types. For all analyses of birds, I excluded individuals flying over, waterfowl, raptors, species detected fewer than 3 times and outliers determined using outlier analysis in PC-ORD v.4.0 (McCune and Mefford 1999).

2.4.2.2 Species Richness

To assess relative species richness of riparian/wetland obligate (hereafter, riparian), canopy, cavity and ground nesting guilds amongst treatments, I used rarefaction estimates produced in EstimateS v.8.0 (Colwell 1997). Guilds were determined by species accounts documented in the Birds of North America (Poole 2005), personal observations and unpublished data (Appendix A3). Rarefaction estimates are produced by repeated random sampling of the data set, plotting the average number of species represented by the number of samples with each simulation (Gotelli and Colwell 2001). The response variable of a rarefaction curve is proportional to the richness of a treatment. When an asymptote on the curve is reached, the curve for that treatment corresponds to its highest richness.

2.4.2.3 Relative Abundance and Frequency of Occurrence

Relative abundance of riparian, upland canopy, upland ground and cavity nesting species between treatments was assessed using ANOVA. I compared distributions of individual species across treatments and disturbance types using Indicator Species

Analysis (ISA; Dufrene and Legendre 1997). This method combines individual species' abundances and frequency of occurrence to derive indicator values for each species in each treatment. Results are contrasted using a Monte Carlo randomization test to determine significance of species' occurrences (McCune and Grace 2002).

2.4.2.4 Bird Community Analysis

To examine bird community responses to disturbance types and post-disturbance vegetation variables, Redundancy Analysis (RDA) was used. RDA is similar to Canonical Correspondence Analysis (CCA) but is based on a linear model of species' responses (ter Braak 1986, 1994). Treatment types were entered in the vegetation matrix as dummy variables, significant pre-disturbance GIS data (from ANOVA) were entered as covariates and post-disturbance vegetation components were included as continuous explanatory variables. To account for a large spatial gradient, a centroid of each wetland represented as longitude was included in the ordination (Legendre 1993). I created two ordinations by running RDA for all (riparian and upland) species and again, separately, for all riparian species in CANOCO v.4.54 (ter Braak and Smilauer 1999). Species abundance data were square-root transformed prior to analysis. Assessment of the variance inflation factors in the correlation matrix was used to assess collinearity of environmental variables (Quinn and Keough 2004). The ordinations were symmetrically scaled and centered by species. To aid in the interpretation of the ordinations and to assess differences in community composition between treatments, 67% confidence ellipses (CE) around treatment groups were used (Hobson and Schieck 1999). A non-parametric test, Multi-Response Permutation Procedures (MRPP) using the Euclidean

Distance measure, was used to determine if differences between treatment groups in the ordinations were significant (McCune and Grace 2002).

2.5 RESULTS

2.5.1 Vegetation

Broken-stick eigenvalues suggested that four axes would most appropriately represent the variability in pre-disturbance vegetation data. As indicated by their scores, Axes 1 and 3 represented 'other' softwood (see Table2.3), Axis 2 represented graminoid vegetation and Axis 4 represented hardwood forest types. These four axes encompassed 81.9% of the cumulative variance in the pre-disturbance vegetation data. Areas of low merchantable softwood (Axis 1; $F_{3,88}= 5.06$, $p<0.01$), graminoid vegetation (Axis 2; $F_{3,88}= 8.64$, $p<0.001$) and hardwood (Axis 4; $F_{3,88}= 4.48$, $p<.01$) differed significantly between treatments in the pre-disturbance data between the non-merchantable and merchantable treatments.

Analysis of post-disturbance vegetation variables derived from GIS showed the first three ordination axes sufficiently described variation in vegetation parameters. As indicated by their PCA scores, Axis 1 represented area disturbed, Axis 2 represented graminoid vegetation and Axis 3 represented area of fen. Cumulatively, these three axes represented 77.6% of the variance in the post-disturbance vegetation data. Additionally, analysis of field data indicated that three axes explained 57.6% of overall variance: Axis 1 was associated with canopy closure (and canopy height); Axis 2 with upland grass cover, and; Axis 3 with canopy height. PCA of the wetland classification data showed

two axes (Axis 1- meadow marsh and Axis 2- treed bog) represented 55.0% of the cumulative variance.

Tests for differences between PCA scores of vegetation variables in each of the data types showed significant differences for seven of eight variables. For GIS data, area disturbed ($F_{3, 88}$= 6.97, p<0.001), graminoid ($F_{3, 88}$= 3.82, p<0.05) and fen ($F_{3, 88}$= 8.99, p<0.001) showed significant between-treatment differences. From field data, significant differences resulted between treatments for canopy closure ($F_{3, 72}$= 54.26, p<0.001), non-riparian grass cover ($F_{3, 72}$= 5.06, p<0.005) and canopy height ($F_{3, 72}$= 4.72, p<0.01). Significant differences were also found between treatments in percent of post-disturbance meadow marsh ($F_{3, 81}$= 11.56, p<0.001).

2.5.2 Avian Analysis

2.5.2.1 Detection Probability Estimation

Detection probability estimates were derived from observations of 8071 individuals representing 112 species at 314 point-count stations in riparian habitats distributed across the Boreal Plain of Alberta, Saskatchewan and Manitoba. Average species- or group-specific detectability estimates ranged from 62% (Black-capped Chickadee *Poecile atricapilla* group; Appendix A2) to 92% (Common Yellowthroat *Geothlypis trichas*) with individual estimates ranging from 51% (Black-capped Chickadee) to 99% (Ovenbird *Seiurus aurocapillus*). The parameters encompassing the greatest proportion of the weights for all species combined were observer ability (14.0%), distance from observer (10.8%), time of day (9.9%) and sampling date (8.3%).

2.5.2.2 Bird Community Analysis

A total of 4525 individuals were detected, representing 131 bird species. All analyses of birds were limited to individuals detected within 50 m of the point count or transect to maximize my ability to make inferences about birds inhabiting riparian areas and disturbances close to the riparian habitat.

2.5.2.3 Species Richness

Thirty meter buffers had the highest overall richness (69 species; Figure 2.3). Estimates of richness in all other treatments for all species are similar with the 10 m buffers having the lowest overall richness (56 species). Highest richness of riparian species was observed in burned non-merchantable sites with 26 species followed closely by burned merchantable and 30 m buffer treatments both with 21 species. Lowest richness in the riparian guild was seen in 10 m buffer sites with 19 species. The 10 m buffer treatment rarefaction curve did not reach an asymptote for this guild.

Highest richness of upland canopy-nesting species was reached in the 30 m buffer treatment at 25 species. Richness of species in this guild was next highest in burned merchantable and 10 m buffer treatments and reached a plateau at 17 species. Richness of cavity-nesters was highest in burned merchantable sites at a maximum richness of 19 species. Rarefaction curves for all other treatments did not reach asymptotes with similar richness estimates between 11 and 13 species. Richness of upland ground-nesting species was similar between all treatments with burned non-merchantable and 10 m buffers both reaching maximum richness at nine species and the other two treatments attaining maximum richness at 10 species.

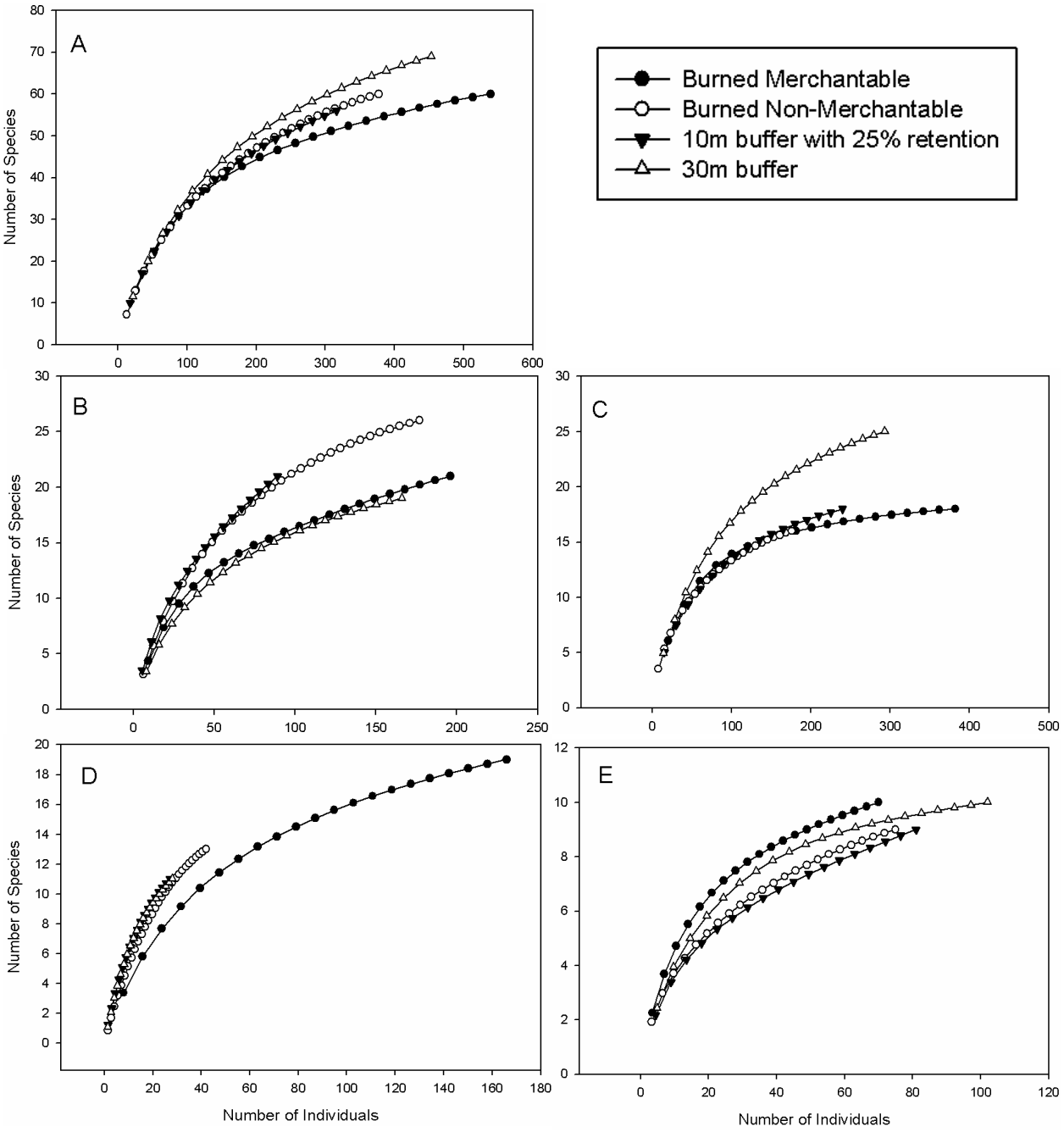

Figure 2.3. Rarefaction curves for: a) all species, b) riparian, c) canopy, d) cavity, and, e) upland ground-nesting guilds in four treatments: 1) Burned merchantable (n= 21), 2) Burned non-merchantable (n= 29), 3) 10 m buffer with 25% retention in the next 30 m (n=19), and, 4) 30 m buffer (n= 21) in the Boreal Plain ecozone of Saskatchewan and Alberta, 2004 and 2005.

2.5.2.4 Total Species and Guild Abundance

Total species abundance was highly variable but differed significantly between treatments ($F_{3, 88}$= 5.82, p<0.001), namely between post-fire sites and between burned non-merchantable and 30 m buffers (Bonferroni post-hoc test, p<0.05). Abundance of all species combined was highest in burned merchantable sites followed by 30 m buffer, 10 m buffer and, lastly, by burned non-merchantable treatments (Figure 2.4a). Abundance of riparian species did not differ between treatments ($F_{3, 86}$= 1.57, p>0.2; Figure 2.4b); however, highest species' abundance was reached in the burned merchantable treatment followed by the 30 m buffer, burned non-merchantable and 10 m buffer. Upland canopy-nesting birds showed significant differences between the burned treatments (Bonferroni post-hoc tests, p<0.01), with highest abundance occurring in burned merchantable sites followed by the two harvested treatments and lastly by burned non-merchantable sites (Figure 2.4c). Cavity nesting species showed non-significant differences between treatments. The burned merchantable treatment possessed the highest abundance of cavity-nesting species with lower mean abundance in all other treatments (Figure 2.4d). I found no significant differences between treatments of upland ground-nesting birds ($F_{3, 81}$= 1.40, p>0.2; Figure 2.4e). Abundance of birds in this guild was highest in 30 m buffers and lowest in burned treatments. Standard deviations were used in Figure 2.4 to show variability of abundance around the mean.

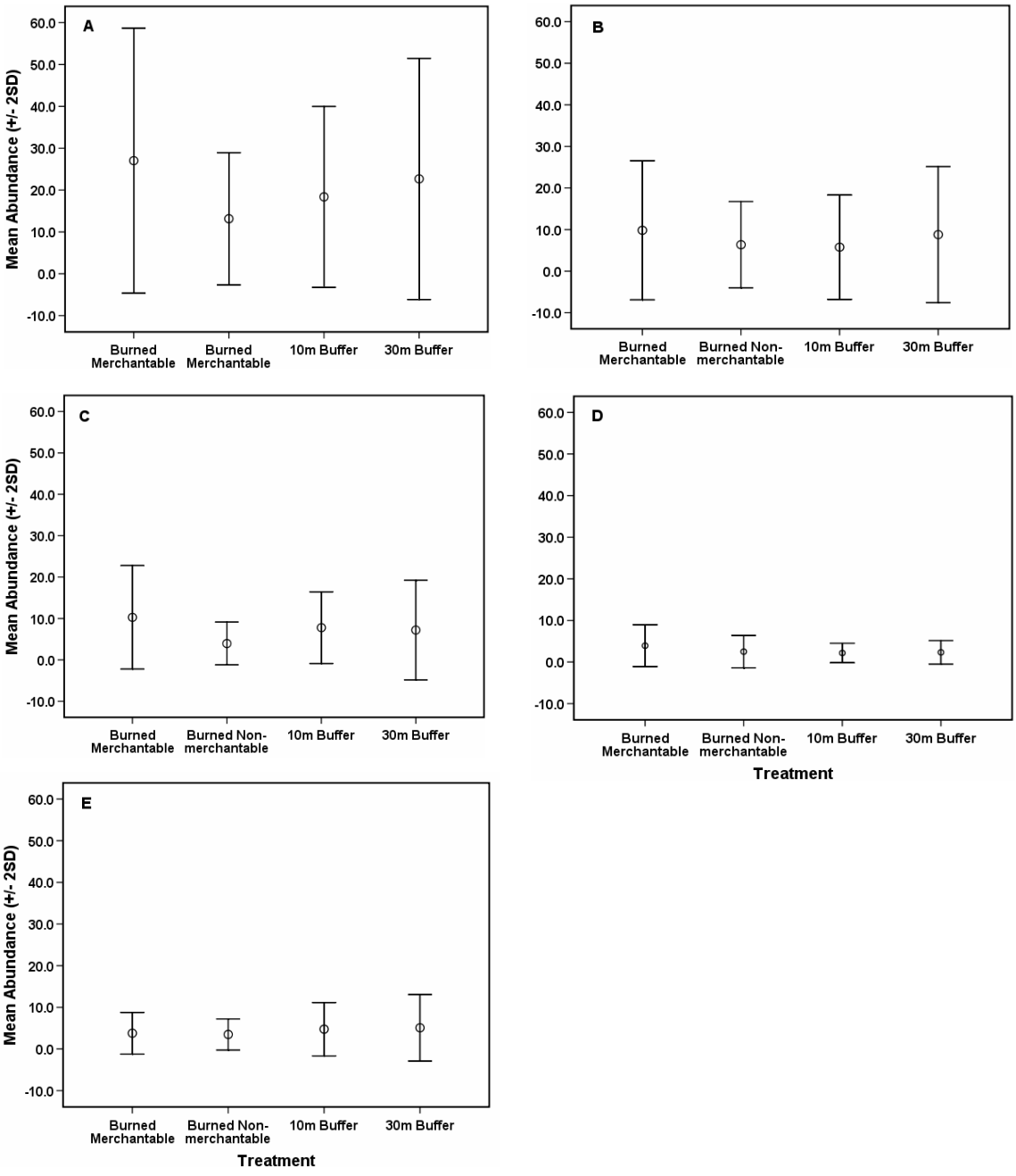

Figure 2.4. Mean abundance (± 2SD) for: a) all species, b) riparian, c) canopy, d) cavity, and, e) ground-nesting species in: 1) burned merchantable (n=21), 2) burned non-merchantable (n=29), 3) 10 m buffer (25% retention in the next 30 m; n= 18), and, 4) 30 m buffers (n=21) in the Boreal Plain of Saskatchewan and Alberta.

2.5.2.5 Indicator Species Analysis

Twenty-three species differed in relative abundance and frequency of occurrence between treatments using Indicator Species Analysis ($p<0.1$; Table 2.4). Ten species were indicative of burned merchantable sites, one of burned non-merchantable, four of 10 m buffers and eight of 30 m buffers. Two riparian species, the Common Yellowthroat and Eastern Kingbird (*Tyrannus tyrannus*) were indicative of burned merchantable sites. The indicator value for Le Conte's Sparrow (*Ammodramus leconteii*) was highest in burned non-merchantable sites; however, it also had a high indicator value in burned merchantable sites suggesting it was responding positively to pyrogenic riparian habitats. Species associated with open habitats were common in the burned merchantable treatment including American Robin (*Turdus migratorius*), Cedar Waxwing (*Bombycilla cedrorum*), Western Wood- Pewee (*Contopus sordidulus*) and Brown-headed Cowbird (*Molothrus ater*). Ten-meter buffers had a high number of indicator species generally associated with coniferous habitats (e.g., Cape May Warbler *Dendroica tigrina*, Gray Jay *Perisoreus canadensis*, Pine Siskin). Indicator species of 30 m buffers were dominated by species associated with upland habitats (e.g., Rose-breasted Grosbeak *Pheucticus ludovicianus*, Swainson's Thrush *Catharus guttatus*, Tennessee Warbler *Vermivora peregrina*). The Tennessee Warbler had high indicator values for both harvest treatments indicating an affinity to disturbed sites with green tree retention. Alder Flycatcher (*Empidonax alnorum*) and Wilson's Warbler (*Wilsonia pusilla*) were riparian species indicative of 30 m buffers while no riparian species were indicative of 10 m buffers.

Table 2.4. Species with significantly ($p<0.1$) greater abundance and frequency of occurrence in: 1) burned merchantable (n= 21), 2) burned non-merchantable (n= 29), 3) 10 m buffers (25% retention in the next 30 m; n= 18)), and, 4) 30 m buffers (n= 21) in the Boreal Plain of Saskatchewan and Alberta, 2004 and 2005 using Indicator Species Analysis (ISA). Merchantability was calculated as the percent of a 50 m rectangular, GIS buffer around the transect survey line that was considered merchantable based on digital forest inventories. Scientific names for species are shown in Appendix A3.

		Indicator Value**			
Maximum Group	**Species**	B1	B2	H1	H2
Burned Merchantable	American Robin	**17**	2	2	6
	Brown-headed Cowbird	**30**	4	0	1
	Clay-colored Sparrow	**33**	4	1	1
	Connecticut Warbler	**11**	0	1	0
	Common Yellowthroat*	**21**	1	0	6
	Eastern Kingbird*	**21**	1	2	1
	House Wren	**22**	0	0	0
	Least Flycatcher	**29**	0	4	1
	Philadelphia Vireo	**18**	0	1	1
	Western Wood-peewee	**22**	7	0	1
Burned Non-Merchantable	Le Conte's Sparrow*	15	**19**	0	2
10 m buffer**	Chipping Sparrow	12	7	**29**	12
	Cape May Warbler	0	0	**15**	1
	Gray Jay	11	0	**18**	3
	Pine Siskin	0	0	**12**	7
30 m buffer	Alder Flycatcher*	14	2	4	**22**
	Black-and-White Warbler	0	2	1	**12**
	Chestnut-sided Warbler	2	0	0	**18**
	Magnolia Warbler	0	0	0	**29**
	Rose-breasted Grosbeak	0	0	0	**19**
	Swainson's Thrush	1	1	4	**19**
	Tennessee Warbler	3	1	22	**24**
	Wilson's Warbler*	0	0	1	**20**

* Riparian indicator species.
**B1- Burned merchantable; B2- Burned non-merchantable; H1- 10 m buffer with 25% retention in the next 30 m (Saskatchewan Environment 2006b); H2- 30 m buffer.

2.5.2.6 Overall Community Ordination

Treatments showed considerable separation between burned and harvested sites in the overall Redundancy Analysis ordination with some overlap between post-fire and post-harvest sites based on 67% Confidence Ellipses (CE; Figure 2.5). High overlap was exhibited within the burned treatments and between the harvested treatments. Post-fire riparian bird communities showed high variation in community composition, which was not encompassed by the harvested sites. Both harvest treatments had similar bird community composition, and sites with more retention (30 m buffers) exhibited greater variability in community composition. Results from MRPP confirmed that overall bird communities associated with each treatment were significantly different ($p<0.0001$). Pair-wise comparisons of all treatments using MRPP were significant ($p<0.001$) with smaller but significant differences for the comparison between the two burned treatments ($p<0.05$) and the two harvest treatments ($p<0.05$).

The first four axes of the overall RDA explained 14.2% of the variance in the species matrix. The first two axes explained 27.2 and 20.5%, respectively, of the species-environment relationship, and the first four axes accounted for a total of 70.6% of the explained variance. I interpreted the first axis as representing a gradient from closed to open canopy habitats. Species most highly associated with Axis 1 are Brown-headed Cowbird and Tree Swallow (*Tachycineta bicolor*). Orange-crowned Warbler (*Vermivora celata*) was most negatively associated with Axis 1. Axis 2 was positively associated with area disturbed and negatively associated with hardwood vegetation; hence, I interpret this axis as representing a gradient from high to low disturbance. Longitude was also negatively correlated with Axis 2. Chipping Sparrow (*Spizella passerina*) and Ruby-

crowned Kinglet (*Regulus calendula*) were negatively associated with this axis while House Wren (*Troglodytes aedon*) was most highly associated with this axis.

Riparian species clustered in two main areas of the overall ordination. A total of 16 riparian species have their highest abundances within the 67% CE's of the burned treatments. Riparian species associated with the burned merchantable treatment and graminoid vectors include the Song Sparrow (*Melospiza melodia*), Le Conte's Sparrow, Yellow-headed Blackbird (*Xanthocephalus xanthocephalus*), Lincoln's Sparrow (*Melospiza lincolnii*), Eastern Kingbird, Common Yellowthroat and Common Snipe (*Gallinago delicata*). In the quadrant of the ordination associated with the burned merchantable vector were cavity-nesting birds (House Wren, Tree Swallow), and species typically found in shrubby (Lincoln's Sparrow, Song Sparrow) and open habitats (Clay-colored Sparrow *Spizella pallida*, Savannah Sparrow *Passerculus sandwichensis*, Western Wood-Pewee). Many species associated with this vector also had high indicator values for this treatment from Indicator Species Analysis, a pattern that is consistent with the results of ordination analyses.

Species positively associated with the vectors representing burned non-merchantable habitat and percent disturbed include Greater Yellowlegs (*Tringa melanoleuca*), House Wren and Swamp Sparrow (*Melospiza georgiana*). A weak association was shown with Olive-sided Flycatcher (*Contopus cooperi*) and Yellow-bellied Sapsucker (*Sphyrapicus varius*) with these environmental variables. Fewer species were associated with the burned non-merchantable vector, which is not consistent with estimates of richness for this treatment. Sites with high canopy closure were highly correlated with 10 m buffers. In general, more species considered forest specialists including many warbler species

were within the CE of harvested treatments (Black-throated Green Warbler *Dendroica virens*, Cape May Warbler, etc.). Species typically found in softwood forest types were also associated with the harvested sites including Pine Siskin, Tennessee Warbler and Bay-breasted Warbler (*Dendroica castanea*). Riparian species associated positively with harvested treatments were Wilson's Warbler and Orange-crowned Warbler. Another riparian species, the Northern Waterthrush (*Seiurus noveboracensis*), was weakly associated with canopy closure and hardwood.

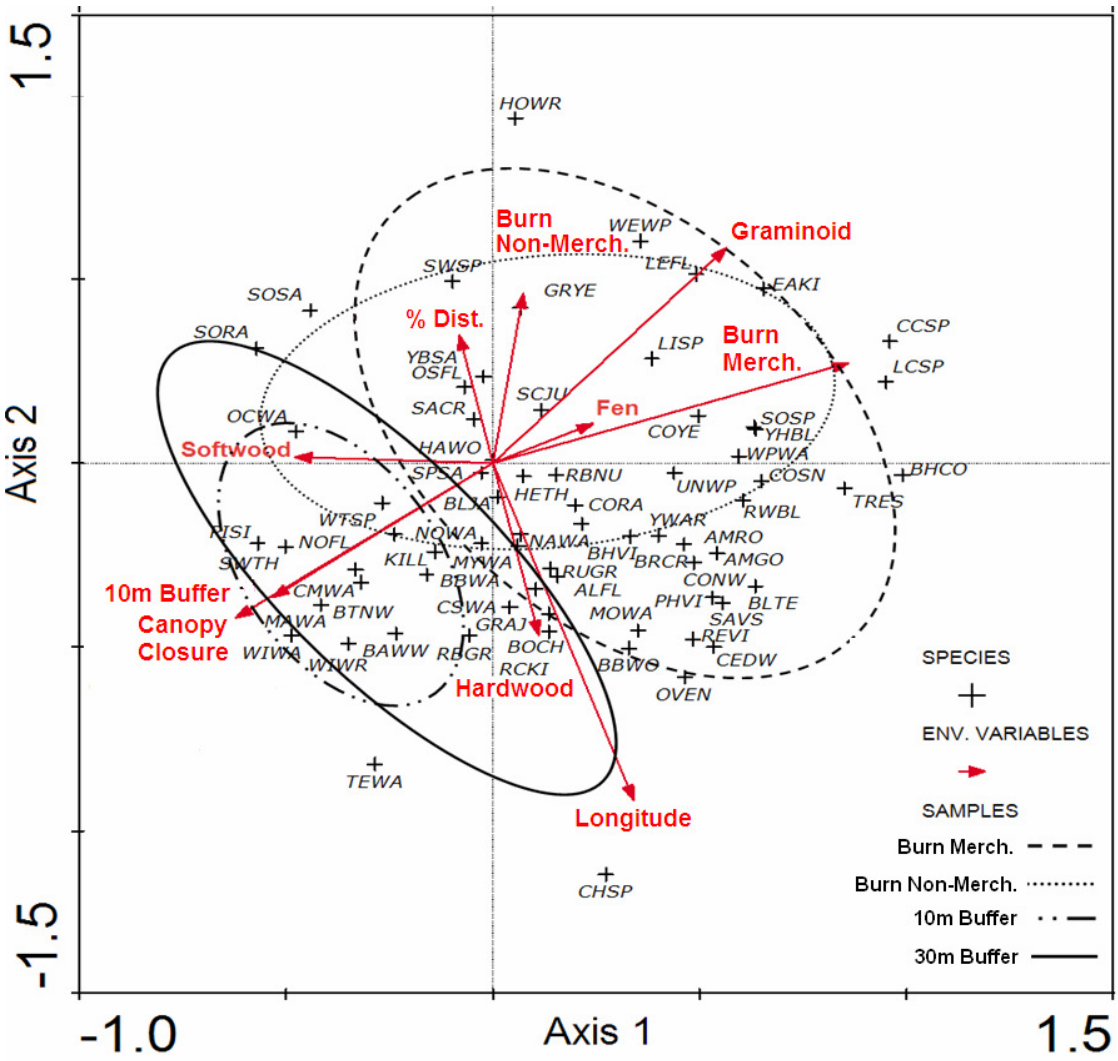

Figure 2.5. Redundancy Analysis (RDA) ordination plot with 67% Confidence Ellipses (CE) for all bird species (71 species) in four treatments: 1) Burned merchantable (n= 21), 2) Burned non-merchantable (n=29), 3) 10 m buffer (25% retention in the next 30 m; n= 18), and, 4) 30 m buffer (n= 21) around small wetlands in the Boreal Plain of Saskatchewan and Alberta, 2004 and 2005. Species codes and scientific names for species are listed in Appendix A3.

2.5.2.7 Wetland/Riparian Species Ordination

RDA for riparian species showed 67% CE around samples in each treatment with greater overlap of treatments compared to the overall ordination (Figure 2.6). High overlap existed between post-fire sites and between post-harvest sites. Ten meter buffers showed a tighter cluster in the ordination being positioned mostly within the CE of 30 m buffers. Results from MRPP, however, showed significant differences between bird communities associated with each treatment ($p<0.001$). Pair-wise comparisons of treatments indicated that only the burned non-merchantable sites and the 10 m buffers had significantly different bird communities ($p<0.01$).

The first two axes of the RDA represented 19.8% of variance in the species data and 59.4% of the species-environment matrix. The first axis of the riparian species data encompassed 7.4% of the total variance in the species data and 39.4% of the total variance in the species-environment relationship. This axis was positively associated with graminoid vegetation, which was positively correlated with the burned merchantable treatment. Canopy closure and softwood was negatively associated with Axis 1. Similar to the overall ordination, I interpreted the first axis as representing a gradient from upland to lowland habitat types. A total of seven riparian species was found exclusively in the 67% CE of the burned treatments and three were found exclusively in the harvest CE. Species showing a strong positive association with Axis 1 were Le Conte's Sparrow, Yellow-headed Blackbird and Eastern Kingbird. Again, Orange-crowned Warbler had the highest negative association with Axis 1.

Axis 2 represented 2.0% of the variance in the bird data and 20.0% of the variance in the species-environment relationship. This axis was positively associated with area of

hardwood vegetation and negatively associated with area of fen; hence, I interpreted this axis as representing a gradient from merchantable to non-merchantable forest types. Burned non-merchantable sites were negatively correlated with this axis and highly correlated with fen. Swamp Sparrow showed the strongest positive association with Axis 2 with Olive-sided Flycatcher and Northern Waterthrush showing weaker associations with this axis. Solitary Sandpiper (*Tringa solitaria*) and Sora Rail (*Porzana carolina*) showed weak, negative associations with Axis 2. Most species showed similar responses to the environmental variables in this analysis as in the overall ordination. Ordination of species in this RDA is consistent with Indicator Species Analysis results, as most species show consistent associations with treatment types.

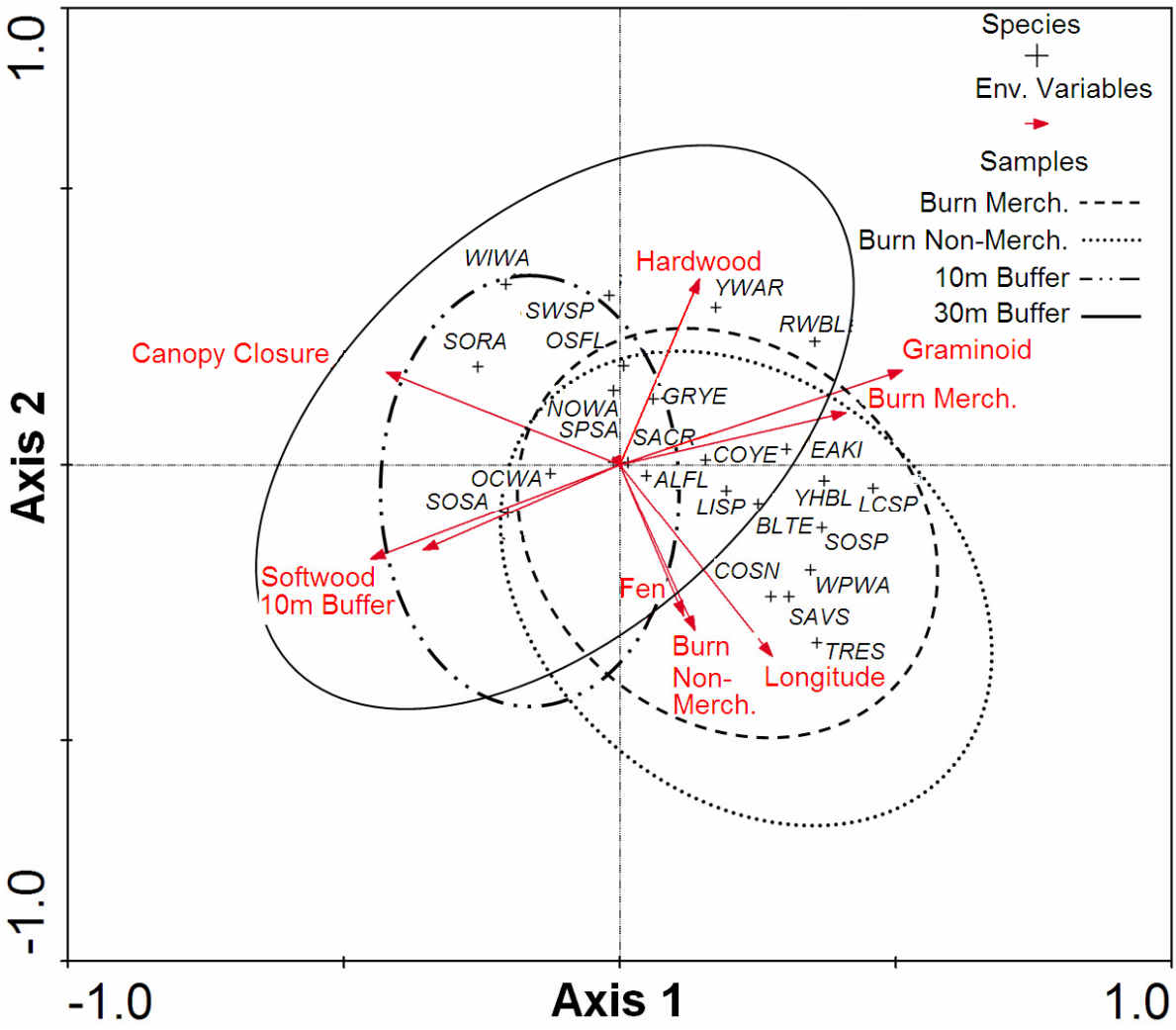

Figure 2.6. Redundancy Analysis (RDA) ordination plot and 67% Confidence Ellipses for riparian-associated species (24 species) in four treatments: 1) Burned merchantable (n= 21), 2) Burned non-merchantable (n= 29), 3) 10 m buffer (25% retention in the next 30 m; n=18), and, 4) 30 m buffer (n= 21) in the Boreal Plain of Saskatchewan and Alberta, 2004 and 2005. Species codes and scientific names are listed in Appendix A3.

2.6 Discussion

While previous studies have investigated the response of upland birds to forestry and fire (Schieck and Song 2006), this study is the first to contrast the influence of natural and anthropogenic disturbances on bird communities at the riparian ecotone in the boreal forest. Although I wasn't able to directly measure vegetation to assess pre-disturbance habitat differences between treatments, digital forest inventory data indicated that I selected sites with similar pre-disturbance shoreline forest and riparian (area of shrub and graminoid) vegetation composition. More detailed measurements of pre-disturbance vegetation, especially in riparian areas and wetlands, would have provided better insight into potential biases in vegetation between treatments. However, use of digital forest inventories represented the only available data for comparison of pre-treatment vegetation in this study and such comparisons are likely correlated with more detailed vegetation parameters (Haeussler and Bergeron 2004). Clear disparities found between post-disturbance vegetation structure and composition of burned versus harvested treatments are consistent with other studies (Hobson and Schieck 1999, Imbeau et al. 1999).

Differences were found in guilds, species abundances and bird community composition between some treatments types. Patterns of high richness of riparian birds observed in early post-fire riparian habitats demonstrate the importance of fire to species inhabiting riparian areas. Presumably, high richness of non-merchantable sites was due to the greater array of habitat types (e.g., bog, fen) surveyed in that treatment. Lower richness and abundance of riparian species in harvested sites, especially those with larger buffers, suggests that lack of vegetation succession may limit riparian bird species in

harvested landscapes (e.g., buffer retention) to the non-forested portion of the riparian habitat by not providing the natural range of habitats similar to post-fire.

The rarefaction curve for the 10m buffer treatment failed to reach an asymptote, which is likely to due to two main factors: 1) poor sampling of rare, irruptive or elusive species (Gotelli and Colwell 2001) and, 2) habitat type. Many riparian species in this study, including Lesser Yellowlegs (*Tringa flavipes*), Sedge Wren (*Cistothorus platensis*), Marsh Wren (*Cistothorus palustris*) and Nelson's Sharp-tailed Sparrow (*Ammodramus nelsoni*) were detected few times (<4 observations). The area of riparian/wetland habitat in the merchantable treatments was low (i.e., in long narrow strips along water) relative to the non-merchantable treatment. Therefore, I was more likely to detect generalist riparian species. Rare species, or those that prefer larger wetland complexes, were only encountered by chance in merchantable sites causing the 10m buffer to not reach an asymptote. Other studies should specifically target rare riparian species (Hannon et al. 2004), especially those that may be sensitive to anthropogenic disturbance or that are considered threatened (i.e., Rusty Blackbird *Euphagus carolinus*; COSEWIC 2006).

Divergence in the overall (upland and riparian) bird community appears to be driven by area of open habitat including canopy closure and graminoid vegetation and in softwood stands. More riparian species were associated with burned treatments including Le Conte's Sparrow, Tree Swallow, Yellow-headed Blackbird, Common Yellowthroat, Palm Warbler, Lincoln's Sparrow and Eastern Kingbird. Early post-fire upland habitats may act as an extension of the riparian ecotone with minimal canopy cover and provide more suitable early successional habitat, allowing some riparian species to expand or

occupy territories protruding into upland habitats. For instance, species such as Common Yellowthroat and Eastern Kingbird detected in regenerating post-fire stands likely responded to more diverse but less-dense shrubby habitats of post-fire environments relative to post-harvest (Schulte and Niemi 1998). The apparent propensity of some species (e.g., Yellow-headed Blackbird) to inhabit burned riparian habitats relative to harvested sites may have more to do with the habitat (i.e., wetland) type or other abiotic factors (e.g., water level, plant structure) than the presence of post-fire features.

However, my results suggest that there were only minimal differences in wetland classification (graminoid and meadow marsh vegetation) between merchantable post-disturbance sites. Other studies (see Schieck and Song 2006) have shown that some riparian species respond positively to post-fire vegetation succession in upland habitats, so burned riparian and shoreline forests may not be critical habitats for riparian species in naturally disturbed areas. Nevertheless, species typically associated with riparian habitats may have greater productivity in burned riparian habitats because of undetectable changes in habitat, differences in predator communities (Imbeau et al. 1999, Fisher and Wilkinson 2005) or in the richness and abundance of prey (McCullough et al. 1998, Buddle et al. 2006) following fire and harvest. Riparian species may also show greater response to abiotic factors including flooding (Lariviere and Lepage 2000), which may be exacerbated in wetlands with disturbed shoreline forests (Van Damme et al. 2003). However, this is speculative and requires further investigation.

Closer convergence of community composition for riparian species than the overall community suggests that the upland bird community is driving differences in the overall community. This result occurred even though I compared burned and unburned riparian

vegetation and suggests less of an impact of fire and shoreline forest harvesting on riparian species versus upland species with harvesting contrasted to fire. This was expected as riparian areas undergo less physical habitat change than does upland forest after disturbance (e.g., killing or removal of trees). Surprisingly, bird communities in larger buffers showed greater overlap with burned sites. Larger buffers may provide a greater range of habitats suitable for more riparian species in the first four years post-harvest. However, smaller buffers may provide habitats more similar to riparian areas in later stages of early succession (5-10 years) with regenerating cutblocks acting as a surrogate for riparian habitat (Hobson and Schieck 1999). Furthermore, harvested treatments that were surveyed did not include any sites that had the entire buffer removed as occurred in most burned sites.

Similar to other studies conducted in the boreal forest (Hobson and Schieck 1999, Imbeau et al. 1999, Hoyt and Hannon 2002), this study found increased richness and abundance of cavity-nesting species in burned habitats. This trend was mainly driven by secondary cavity-nesting species including Tree Swallow, House Wren and Yellow-bellied Sapsucker. The rarefaction curves for cavity-nesting species richness did not reach an asymptote for any of the treatments but the burned merchantable treatment. This was likely because some species have large home ranges (e.g. Pileated Woodpecker *Dryocopus pileatus*) or are rare because they only opportunistically inhabit areas that were surveyed (e.g., Three-toed Woodpecker *Picoides dorsalis*; Hoyt and Hannon 2002).

Species generally associated with open habitats were common in both post-fire (American Robin, Song Sparrow) and post-harvest (Killdeer *Charadrius vociferus*, Magnolia Warbler *Dendroica magnolia*, Northern Flicker *Colaptes auratus*) sites. Higher

richness of canopy-nesting species in harvested sites was contrasted by slightly higher abundance of canopy-nesters in burned merchantable sites. The increased presence of species associated with coniferous forests (e.g., Bay-breasted Warbler, Cape May Warbler) in 10 m buffer sites corresponds to slightly higher amounts of coniferous forest retained in these buffers relative to the 30 m buffer sites. There is also a possibility that differences in pre-disturbance habitat structure, mainly the proportion of hardwoods versus softwoods between harvest treatments, drove the observed bird community response. Between-treatment comparisons of the pre-disturbance digital forest inventory data did not support this but differences using these data may not provide the resolution required to detect such changes.

The Gray Jay, a nest predator, and the Brown-headed Cowbird, a brood parasite, had higher abundance and frequency of occurrence in harvested and burned sites, respectively. The double edge of riparian buffers may be attractive to nest predators because it is easier to find nests in these habitats (Boulet et al. 2003, Ibarzabal and Desrochers 2004, Morgan et al. 2006) compared to contiguous forest. I found higher cowbird abundance in post-fire sites. Cowbirds may prefer burned habitats because they resemble open parkland habitats with less dense shrub growth than post-harvest sites. However, this result seems to be driven by one fire in the southern portion of the study area near agricultural land that contributed 59% of the total detections of cowbirds in all sites and 64% of the burned sites.

I used abundance measures as a proxy for habitat quality which may not be a good indication of habitat suitability or nest success (Van Horne 1983, but see Bock and Jones 2004). Recent studies have shown increased densities of birds occupying territories in

buffers suggesting a 'packing' effect in the year immediately after harvest (Warkentin et al. 2003). Some species inhabiting buffer strips may experience reduced nest success because of higher predation pressure, increased brood parasitism, decreased foraging efficiency and changes in breeding behavior (Hannon et al. 2002, Warkentin et al. 2003, Tewksbury et al. 2006). Further studies should assess nest success of riparian species in post-fire and post-harvest riparian habitats and determine inter- and intra-year variation of bird species and communities inhabiting riparian areas because these areas may undergo high temporal flooding and temperature fluctuations. Such information is sparse and would be useful in understanding the variability of bird communities in response to such factors. This data would provide a more accurate baseline for comparisons of bird communities in anthropogenically-disturbed forests.

2.7 Management Implications

Natural disturbances are critical components of the ecology of the boreal forest and to the maintenance of the species that inhabit them (Potter and Kessell 1980, Brawn et al. 2001). Current shoreline forest harvesting guidelines were not originally developed to approximate natural disturbances and management policies typically mandate the retention of one-sized treed buffers to mitigate the potential negative impacts of forestry on water bodies (Lee et al. 2004). This study has shown that fire appears to be an important component of the disturbance regime in boreal shoreline forests and riparian areas for birds. Overall, bird communities in post-fire riparian sites exhibit a high natural range of variation. Forest managers attempting to maintain natural diversity and richness on the landscape should consider bird community responses to fire and buffer treatments.

The retention of larger (30 m) treed buffer strips may be a management scenario suitable for some species, especially those that prefer tall and dense shrub habitats with some canopy cover (e.g., Alder Flycatcher, Northern Waterthrush). However, a static approach to forest management may not satisfy the needs of all riparian species. Some species, including Common Yellowthroat and Lincoln Sparrow were detected in regenerating post-harvest (1-5 years) uplands. As such, use of buffer strips across the landscape may not provide habitat for all riparian species especially if productivity of these species is higher at the riparian-upland interface than in regenerating upland forests. Riparian community analysis showed less divergence than the combined upland and riparian analysis. So, species inhabiting riparian areas may be less impacted by disturbances in the shoreline forest and riparian habitat than upland bird communities. While lack of a major response to disturbance type by riparian species does not justify eliminating buffers on the landscape, it does suggest the exploration of alternative management scenarios for shoreline forests.

Forest harvesting is expected to truncate forest age distribution (~70 years) with the majority of old-growth forest remaining in buffer strips. Therefore, species that may be most at risk of population declines due to forest harvesting are those that require old-growth forest (e.g., Black-throated Green Warbler). These species may benefit from forestry practices that take into account the retention of old-growth in larger patches (Schmiegelow et al. 1997). One way to achieve this is by re-allocating trees in buffer strips to larger aggregate patches. This approach would require management at a landscape level that includes incorporating riparian areas to ensure that habitat preferences of all species on the landscape are adhered to, including riparian species. The

trade-off between managing buffers as static elements of the landscape as opposed to dispersing the same volume of timber as residual in other portions of the landscape should be further assessed, potentially via simulation studies (e.g., Rempel and Kaufmann 2003).

Using a flexible template for harvesting shoreline forests based on the Natural Disturbance Paradigm may be a step closer to approximating natural disturbances for riparian birds. This may include eliminating buffers or applying various widths of buffers on single wetlands or on different wetlands across the landscape to encompass the natural range of variability of birds in post-fire environments. This is speculative and more research is needed to investigate early post-fire riparian and shoreline forest vegetation composition and structure. Such research should quantifying residual patch size and composition at the riparian edge with different shoreline forest types and wetland types and bird responses to these variables relative to similar configurations in harvested areas.

Another scenario that warrants further research, but is likely to be less preferred by harvesting companies, is to burn buffer strips. This would create habitats in harvested landscapes most similar to naturally burned shoreline forests and riparian sites. While the idea of prescribing fires in the boreal forest is not likely to garner much support, fires would be much easier to control in riparian buffers than in upland harvests. However, if riparian buffers represent the only remaining old-growth on the landscape, this is likely not a suitable option. Consequences of eliminating fire from the landscape on forest structure and productivity in the long term is unknown but could result in more dramatic shifts in bird community dynamics in riparian habitats.

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Appendix A1. Models evaluated for relative plausibility in estimating detection probabilities of birds around small wetlands in the Boreal Plain of Saskatchewan, Alberta and Manitoba using Huggins' closed captures. Vegetation parameters were extracted from 150 m radii buffers on point count stations using GIS and represent the proportion of a 50 m rectangular buffer around the transect survey line.

1) Effort
2) Effort + Time
3) Effort + Time + Time^2
4) Effort + Time + Time^2 + Time^3
5) Effort + Date + Date^2
6) Effort + Date + Date^2+Date^3
7) Effort + Date + Time + Date*Hours
8) Effort + High Observer Ability + Moderate Observer Ability
9) Effort + Distance
10) Effort + % Deciduous
11) Effort + % Coniferous
12) Effort + % Open
13) Effort + % Cut
14) Effort + % Burn
15) Effort + % Burn + % Deciduous
16) Effort + % Burn + % Open
17) Effort + % Deciduous + % Open
18) Effort + % Deciduous + % Open + % Deciduous* % Open
19) Effort + % Deciduous + % Cut
20) Effort + % Deciduous + % Cut + % Deciduous* % Cut
21) Effort + % Deciduous + % Open + % Cut
22) Effort + % Deciduous + % Open + % Cut + % Deciduous* % Open + % Deciduous*% Cut + % Open* % Cut
23) Effort + % Open + % Cut
24) Effort + Distance + % Open + % Cut
25) Effort + Distance + % Open
26) Effort + Distance + % Open + Distance* % Open
27) Effort + Distance + % Deciduous
28) Effort + Distance + % Deciduous + % Distance* % Deciduous

*Effort represents Ln(10) to account for unequal probability of detecting birds in successive time intervals. Time is when the survey was completed. Distance is the midpoint of the distance band bird was detected in (0-10, 10-25, 25-50, 50-100 and >100 m). Observer ability is a rank of Open represents a combination of graminoid and shrub vegetation.

Appendix A2. Number of detections (n), average detection probability estimates (DPE ± 1SE) and minimum and maximum detection estimates for species or groups of species encountered during 10-minute point counts around small wetlands in the Boreal Plain of Saskatchewan, Alberta and Manitoba 2004-2006. Scientific names for species are shown in Appendix A3.

Species and Groups	n	Average DPE (%) ± 1SE	Min (%)	Max (%)
Alder Flycatcher/Olive-sided Flycatcher/Yellow-bellied Flycatcher	175	81.94 ± 0.30	68.53	95.46
American Redstart	236	78.46 ± 0.85	54.79	94.65
American Robin/Western Tanager	182	79.31 ± 0.79	52.45	94.48
Bay-breasted Warbler/Cape May Warbler/Myrtle Warbler	173	79.81 ± 0.47	51.17	90.49
Black-capped Chickadee/Boreal Chickadee/Red-breasted Nuthatch	103	61.74±1.59	50.58	84.37
Blackburnian Warbler/Black-throated Green Warbler	87	69.35 ± 0.33	63.49	73.55
Blue Jay/Gray Jay	128	69.94 ± 0.51	59.23	86.99
Brown Creeper	72	73.55 ± 1.19	60.69	91.01
Brown-headed Cowbird	116	75.18 ± 1.80	51.16	93.30
Canada Warbler/Magnolia Warbler	81	80.29 ± 0.55	58.31	93.56
Cedar Waxwing/American Goldfinch	103	76.15 ± 0.79	63.30	95.09
Chipping Sparrow/Dark-eyed Junco	307	77.87 ± 0.05	60.47	84.26
Common Raven/American Crow	167	75.60 ± 0.27	64.27	83.25
Common Snipe	103	82.68 ± 1.15	68.69	99.99
Common Yellowthroat	342	91.57 ± 0.31	82.47	97.50
Chestnut-sided Warbler/Yellow Warbler	384	78.92 ± 0.77	60.34	93.82
Least Flycatcher	145	88.78 ± 0.18	85.15	93.80
Le Conte's Sparrow/Savannah Sparrow/Black-and-white Warbler	102	81.21± 0.34	72.79	89.30
Lincoln's Sparrow	75	83.40 ± 0.89	64.52	96.95
Mourning Warbler/Connecticut Warbler	211	84.84 ± 0.18	82.11	91.81
Northern Waterthrush	153	75.41 ± 0.11	72.56	79.68
Ovenbird	308	88.56 ± 0.49	70.10	99.00

Appendix A2 (continued).

Species and Groups	n	Average DPE (%) ± 1SE	Min(%)	Max(%)
Red-eyed Vireo/Blue-headed Vireo/Philadelphia Vireo	480	85.27 ± 0.08	79.61	89.18
Red-winged Blackbird/Yellow-headed Blackbird/Common Grackle	402	79.04 ± 0.34	65.68	93.82
Rose-breasted Grosbeak	141	68.23 ± 0.61	52.27	86.21
Ruby-crowned Kinglet/Golden-crowned Kinglet	128	61.74 ± 0.57	58.91	96.45
Song Sparrow	228	77.63 ± 0.35	60.51	91.14
Spotted Sandpiper/Solitary Sandpiper/Greater Yellowlegs/Lesser Yellowlegs/Killdeer	93	76.34 ± 0.30	68.80	84.93
Swainson's Thrush/Hermit Thrush	135	80.12 ± 0.43	67.00	90.87
Swamp Sparrow/Clay-colored Sparrow	525	80.60 ± 0.15	72.16	85.62
Tennessee Warbler/Nashville Warbler	189	80.12 ± 0.62	68.21	98.03
Western Wood-pewee/Eastern Phoebe	81	91.06 ± 0.33	85.47	96.45
Winter Wren/House Wren	85	79.83 ± 0.30	73.96	84.57
White-throated Sparrow	970	79.81 ± 0.17	74.31	97.34
Woodpeckers- Black-backed/Three-toed/Pileated/Hairy/Downy/ Northern Flicker/Yellow-bellied Sapsucker	153	77.85 ± 0.50	58.99	88.05

*Species were grouped where individual species detections were <80. Groupings were based on song characteristics, singing frequency, behavioral traits and habitat preferences (Alldredge et al. 2007).

Appendix A3. Common and scientific names, nesting guilds, species abbreviations and mean abundance (± 1SE) of species with greater than three detections in four treatments: 1) Burned merchantable (n= 21), 2) Burned non-merchantable (n= 29), 3) 10 m buffer with 25% retention in the next 30 m (n= 18), and, 4) 30 m buffer (n= 21) in the Boreal Plain of Saskatchewan and Alberta.

Species	Nesting Guild	Code	Burned Merchantable Mean ± 1SE	Burned Non-Merchantable Mean ± 1SE	10 m Buffer Mean ± 1SE	30 m Buffer Mean ± 1SE
Alder Flycatcher* *Empidonax alnorum*	S	ALFL	0.99 ± 0.30	0.27 ± 0.11	0.53 ± 0.26	1.25 ± 0.39
American Goldfinch *Carduelis tristis*	T	AMGO	0.12 ± 0.08	0.16 ± 0.12	0.00 ± 0.00	0.27 ± 0.17
American Redstart *Setophaga ruticilla*	T	AMRE	0.00 ± 0.00	0.07 ± 0.07	0.06 ± 0.06	0.16 ± 0.16
American Robin *Turdus migratorius*	T	AMRO	0.61 ± 0.20	0.17 ± 0.07	0.18 ± 0.10	0.40 ± 0.22
Black-and-white Warbler *Mniotilta varia*	Gr	BAWW	0.00 ± 0.00	0.08 ± 0.06	0.06 ± 0.06	0.22 ± 0.10
Black-backed Woodpecker *Picoides arcticus*	C	BBWO	0.14 ± 0.14	0.08 ± 0.06	0.00 ± 0.00	0.00 ± 0.00
Brown-headed Cowbird *Molothrus ater*	T	BHCO	0.85 ± 0.23	0.27 ± 0.15	0.00 ± 0.00	0.10 ± 0.07
Black Tern* *Chlidonias niger*	F	BLTE	0.24 ± 0.19	0.03 ± 0.03	0.00 ± 0.00	0.00 ± 0.00
Black-throated Green Warbler *Dendroica virens*	T	BTNW	0.00 ± 0.00	0.05 ± 0.05	0.00 ± 0.00	0.19 ± 0.15
Blue-headed Vireo *Vireo solitarius*	T	BHVI	0.14 ± 0.10	0.00 ± 0.00	0.11 ± 0.08	0.05 ± 0.05
Boreal Chickadee *Poecile hudsonicus*	C	BOCH	0.42 ± 0.22	0.05 ± 0.05	0.16 ± 0.12	0.25 ± 0.15
Brown Creeper *Certhia Americana*	C	BRCR	0.39 ± 0.26	0.09 ± 0.09	0.14 ± 0.10	0.07 ± 0.07

Appendix A3 (continued).

Species	Nesting Guild	Code	Burned Merchantable Mean ± 1SE	Burned Non-Merchantable Mean ± 1SE	10 m Buffer Mean ± 1SE	30 m Buffer Mean ± 1SE
Cape May Warbler *Dendroica tigrina*	T	CMWA	0.00 ± 0.00	0.00 ± 0.00	0.34 ± 0.22	0.05 ± 0.05
Chestnut-sided Warbler *Dendroica pensylvanica*	S	CSWA	0.15 ± 0.11	0.03 ± 0.03	0.06 ± 0.06	0.41 ± 0.16
Chipping Sparrow *Spizella passerine*	T	CHSP	1.63 ± 0.53	0.95 ± 0.24	2.31 ± 0.51	1.39 ± 0.39
Clay-colored Sparrow *Spizella pallida*	S	CCSP	1.40 ± 0.40	0.45 ± 0.19	0.23 ± 0.18	0.14 ± 0.10
Common Raven *Corvus corax*	T	CORA	0.14 ± 0.08	0.04 ± 0.04	0.09 ± 0.09	0.10 ± 0.10
Common Snipe* *Gallinago delicata*	Gr	COSN	0.13 ± 0.09	0.27 ± 0.12	0.00 ± 0.00	0.05 ± 0.05
Common Yellowthroat* *Geothlypis trichas*	Gr	COYE	0.83 ± 0.29	0.14 ± 0.08	0.11 ± 0.11	0.44 ± 0.22
Connecticut Warbler *Oporornis agilis*	Gr	CONW	0.21 ± 0.12	0.00 ± 0.00	0.07 ± 0.07	0.00 ± 0.00
Dark-eyed Junco *Junco hyemalis*	Gr	DEJU	0.39 ± 0.14	0.77 ± 0.23	0.25 ± 0.14	0.22 ± 0.10
Eastern Kingbird* *Tyrannus tyrannus*	T	EAKI	0.57 ± 0.20	0.14 ± 0.09	0.19 ± 0.15	0.11 ± 0.11
Gray Jay *Perisoreus canadensis*	T	GRAJ	0.57 ± 0.22	0.00 ± 0.00	0.66 ± 0.22	0.22 ± 0.10
Greater Yellowlegs* *Tringa melanoleuca*	Gr	GRYE	0.13 ± 0.09	0.16 ± 0.09	0.13 ± 0.13	0.12 ± 0.09
Hairy Woodpecker *Picoides villosus*	T	HAWO	0.11 ± 0.08	0.03 ± 0.03	0.07 ± 0.07	0.11 ± 0.08
Hermit Thrush *Catharus guttatus*	Gr	HETH	0.00 ± 0.00	0.14 ± 0.08	0.06 ± 0.06	0.00 ± 0.00

Appendix A3 (continued).

Species	Nesting Guild	Code	Burned Merchantable Mean ± 1SE	Burned Non-Merchantable Mean ± 1SE	10 m Buffer Mean ± 1SE	30 m Buffer Mean ± 1SE
House Wren *Troglodytes aedon*	C	HOWR	0.49 ± 0.20	0.03 ± 0.03	0.06 ± 0.06	0.06 ± 0.06
Killdeer* *Charadrius vociferus*	Gr	KILL	0.10 ± 0.07	0.00 ± 0.00	0.17 ± 0.09	0.33 ± 0.19
Le Conte's Sparrow* *Ammodramus leconteii*	Gr	LCSP	0.73 ± 0.21	1.23 ± 0.42	0.06 ± 0.06	0.31 ± 0.20
Least Flycatcher *Empidonax minimus*	T	LEFL	1.64 ± 0.57	0.07 ± 0.05	0.48 ± 0.23	0.26 ± 0.18
Lincoln's Sparrow* *Melospiza lincolnii*	Gr	LISP	0.83 ± 0.38	0.45 ± 0.17	0.30 ± 0.15	0.20 ± 0.09
Magnolia Warbler *Dendroica magnolia*	T	MAWA	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.51 ± 0.19
Mourning Warbler *Oporornis philadelphia*	Gr	MOWA	0.60 ± 0.19	0.00 ± 0.00	0.38 ± 0.13	0.61 ± 0.27
Nashville Warbler *Vermivora ruficapilla*	Gr	NAWA	0.05 ± 0.05	0.07 ± 0.07	0.13 ± 0.13	0.10 ± 0.10
Northern Flicker *Colaptes auratus*	C	NOFL	0.00 ± 0.00	0.07 ± 0.05	0.11 ± 0.08	0.19 ± 0.11
Northern Waterthrush* *Seiurus noveboracensis*	Gr	NOWA	0.06 ± 0.06	0.08 ± 0.08	0.17 ± 0.17	0.37 ± 0.20
Olive-sided Flycatcher* *Contopus cooperi*	T	OSFL	0.00 ± 0.00	0.13 ± 0.09	0.06 ± 0.06	0.05 ± 0.05
Ovenbird *Seiurus aurocapillus*	Gr	OVEN	0.30 ± 0.15	0.03 ± 0.03	0.00 ± 0.00	0.34 ± 0.18
Palm Warbler* *Dendroica palmarum*	Gr	YBHL	0.15 ± 0.15	0.07 ± 0.05	0.00 ± 0.00	0.05 ± 0.05
Philadelphia Vireo *Vireo philadelphicus*	T	PHVI	0.31 ± 0.11	0.00 ± 0.00	0.12 ± 0.12	0.06 ± 0.06

Appendix A3 (continued).

Species	Nesting Guild	Code	Burned Merchantable Mean ± 1SE	Burned Non-Merchantable Mean ± 1SE	10 m Buffer Mean ± 1SE	30 m Buffer Mean ± 1SE
Pine Siskin *Carduelis pinus*	T	PISI	0.00 ± 0.00	0.00 ± 0.00	0.44 ± 0.23	0.38 ± 0.29
Ruby-crowned Kinglet *Regulus calendula*	T	RCKI	0.42 ± 0.21	0.12 ± 0.08	0.3 ± 0.19	0.33 ± 0.14
Red-breasted Nuthatch *Sitta canadensis*	C	RBNU	0.05 ± 0.05	0.07 ± 0.07	0.06 ± 0.06	0.05 ± 0.05
Red-eyed Vireo *Vireo olivaceus*	T	REVI	0.32 ± 0.11	0.11 ± 0.09	0.06 ± 0.06	0.62 ± 0.28
Red-winged Blackbird* *Agelaius phoeniceus*	Em	RWBL	1.81 ± 0.62	0.96 ± 0.43	0.34 ± 0.12	2.29 ± 1.24
Rose-breasted Grosbeak *Pheucticus ludovicianus*	T	RBGR	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.26 ± 0.14
Sandhill Crane* *Grus canadensis*	F	SACR	0.00 ± 0.00	0.10 ± 0.07	0.06 ± 0.06	0.05 ± 0.05
Savannah Sparrow *Passerculus sandwichensis*	Gr	SAVS	0.05 ± 0.05	0.08 ± 0.06	0.00 ± 0.00	0.10 ± 0.07
Sora* *Porzana carolina*	Em	SORA	0.19 ± 0.09	0.10 ± 0.06	0.50 ± 0.22	0.36 ± 0.14
Solitary Sandpiper* *Tringa solitaria*	T	SOSA	0.10 ± 0.10	0.17 ± 0.08	0.13 ± 0.09	0.10 ± 0.07
Song Sparrow* *Melospiza melodia*	Gr	SOSP	0.64 ± 0.22	0.29 ± 0.11	0.69 ± 0.25	0.14 ± 0.10
Spotted Sandpiper *Actitis macularius*	Gr	SPSA	0.05 ± 0.05	0.07 ± 0.07	0.00 ± 0.00	0.05 ± 0.05
Swainson's Thrush *Catharus ustulatus*	T	SWTH	0.12 ± 0.09	0.11 ± 0.08	0.26 ± 0.15	0.48 ± 0.14
Swamp Sparrow* *Melospiza georgiana*	Em	SWSP	1.29 ± 0.34	0.82 ± 0.24	1.03 ± 0.28	1.30 ± 0.43

Appendix A3 (continued).

Species	Nesting Guild	Code	Burned Merchantable Mean ± 1SE	Burned Non-Merchantable Mean ± 1SE	10 m Buffer Mean ± 1SE	30 m Buffer Mean ± 1SE
Tennessee Warbler *Vermivora peregrina*	Gr	TEWA	0.12 ± 0.09	0.11 ± 0.08	0.26 ± 0.15	0.48 ± 0.14
Warbling Vireo *Vireo gilvus*	T	WAVI	0.65 ± 0.28	0.52 ± 0.23	0.33 ± 0.18	0.00 ± 0.00
Western Wood-Pewee *Contopus sordidulus*	T	WEWP	0.55 ± 0.17	0.29 ± 0.1	0.00 ± 0.00	0.1 ± 0.07
Wilson's Warbler* *Wilsonia pusilla*	Gr	WIWA	0.00 ± 0.00	0.00 ± 0.00	0.06 ± 0.06	0.33 ± 0.16
Winter Wren *Troglodytes troglodytes*	C	WIWR	0.00 ± 0.00	0.17 ± 0.12	0.33 ± 0.16	0.26 ± 0.13
White-throated Sparrow *Zonotrichia albicollis*	Gr	WTSP	0.05 ± 0.05	0.09 ± 0.09	0.00 ± 0.00	0.06 ± 0.06
Yellow-bellied Sapsucker *Sphyrapicus varius*	T	YBSA	1.37 ± 0.28	1.17 ± 0.24	1.91 ± 0.42	1.20 ± 0.30
Yellow-headed Blackbird* *Xanthocephalus xanthocephalus*	Em	YHBL	0.23 ± 0.13	0.07 ± 0.05	0.18 ± 0.10	0.05 ± 0.05
Yellow-rumped Warbler *Dendroica coronata*	T	MYWA	0.77 ± 0.23	0.45 ± 0.14	0.98 ± 0.28	0.36 ± 0.15
Yellow Warbler* *Dendroica petechia*	S	YWAR	0.88 ± 0.42	0.03 ± 0.03	0.43 ± 0.26	0.67 ± 0.26

*Wetland obligate or riparian associated species

**Nesting guild: C- Cavity; Em-Emergent; Gr- Ground; S- Shrub; T- Tree Canopy

CHAPTER 3. BIRD COMMUNITY CHANGE FOLLOWING HARVESTING IN BOREAL PLAIN RIPARIAN HABITATS OF DUCK MOUNTAIN, MANITOBA: AN EXPERIMENTAL APPROACH

3.1 Abstract

Shoreline forests around small boreal wetlands were subjected to three forest harvest treatments using a Before-and-After Controlled Impact (BACI) study design. Bird communities were surveyed along the riparian interface one year prior to harvest and two years after harvest to determine the response of birds in sites with: 1) 5-35% retention (0 m buffer; n=7); 2) 35-75% retention (10 m buffer with variable retention in the next 30 m; n=10); 3) 75-100% retention (50 m buffer; n=8) and unharvested sites (n=9). Species, guilds and communities showed high variability in their response to harvest treatments. Repeated measures Analysis of Variance (ANOVA) indicated that eight of 22 species responded to treatment, year or year*treatment effects ($p<0.1$) including two riparian species, the Common Yellowthroat (*Geothlypis trichas*) and Song Sparrow (*Melospiza melodia*). Multiple-response Permutation Procedures (MRPP) indicated that similar overall (upland and riparian) pre-disturbance bird communities diverged over time ($p<0.05$) in all harvest treatments whereas riparian bird communities did not. Multivariate regression trees (MRTs) indicated that the overall bird community responded to amount of hardwood/mixedwood forest and amount of harvest while the riparian community responded to amount of graminoid and shrub vegetation. Lack of a detectable negative response from riparian birds suggests that alternative management strategies that incorporate landscape-based planning compared to conventional fixed-width buffer

applications should be explored. This approach would allow management of forests adjacent to riparian areas and in the upland as a single entity, giving greater flexibility for allocating forest from buffers into larger patches adjacent to water bodies or in the upland. However, further monitoring of these post-disturbance communities is now required to determine how robust the riparian communities are through time.

3.2 Introduction

Riparian areas are defined as ecotonal habitats that occur between aquatic and terrestrial environments (Gregory et al. 1991, Naiman et al. 2005). Due to biological, physical, geochemical and hydrological processes at the riparian interface, they are among the most productive and heterogeneous habitats on many landscapes (Gregory et al. 1991, Naiman et al. 1993, Sabo 2005). Apparently in response to these unique landscape features, birds occur in boreal forest riparian areas in greater richness and abundance than the surrounding upland (Whitaker et al. 2000, Macdonald et al. 2006).

Since riparian zones are thought to provide distinct value in terms of conservation, some researchers suggest that riparian areas should receive high conservation priority in industrial landscapes (Spackman and Hughes 1995, Naiman and Decamps 1997, Sarakinos et al. 2001, Decamps et al 2004). Indeed, forest management guidelines require commercial logging operations to retain forested buffer strips adjacent to most wetlands and riparian areas in North American boreal forests (see Lee et al. 2004). Buffer strip retention is intended to minimize potential negative effects of forest harvesting on water quality and fish habitat, to maintain aesthetics and to provide habitat for upland game species (Barten 2001, Lee et al. 2004). Buffer prescriptions vary across Forest

Management Areas (FMAs) in the Boreal Plain ecozone and range from 0 m on ephemeral wetlands to >100 m on lakes with high recreational value (Manitoba Natural Resources 1996, Lee et al. 2004).

Evaluation of avian responses to riparian management strategies has largely been restricted to upland-nesting birds using various widths of buffers retained after harvest. In general, decreases in buffer width lead to declines of forest-dwelling species and overall increases in the density of edge-associated species occupying buffer strips immediately after harvest (Pearson and Manuwal 2001, Hanowski et al. 2005). Buffer widths >100 m appear to meet the requirements of most upland species concentrated along riverine habitats and lakes in mountain ecozones (Kinley and Newhouse 1997, Hagar 1999, Pearson and Manuwal 2001, Shirley and Smith 2005), eastern boreal forests (Darveau et al. 1995, LaRue et al. 1995, Whitaker and Montevecchi 1997, Warkentin et al. 2003), eastern hardwood forests (Meiklejohn and Hughes 1999, Bub et al. 2004, Hanowski et al. 2005) and the Boreal Plain (Lambert and Hannon 2000, Hannon et al. 2002). However, the use of riparian buffer strips as a management strategy presents several potential problems in the conservation of upland-nesting birds. Small buffer strips may act as ecological traps as the dual edge increases the probability of nest predation (Darveau et al. 1997, but see Boulet et al. 2003) or nest parasitism (Howell et al. 2007). Increased densities of birds inhabiting buffers can lead to negative behavioral (e.g., reduced foraging efficiency) and social effects that reduce nest success (Hagan et al. 1996, Warkentin et al. 2003, Lampila et al. 2005). With low or non-existent targets for old-growth forest retention in harvested landscapes, treed buffers may contain the greatest proportion of old-growth forests remaining on the landscape (Lee and Barker 2005). This

approach to the conservation of old-growth forests (e.g., in narrow linear strips) is not effective for bird species that require large tracts of contiguous old-growth habitat (Hobson and Bayne 2000a, b, Hoyt and Hannon 2002, Venier and Pearce 2005).

Boreal riparian areas and the forests adjacent to them are subject to natural disturbances (e.g., fires) at rates equal to or greater than upland forest (Burgess 1997, Harper and Macdonald 2001, Macdonald et al. 2004, 2006, Martell et al. 2006). Therefore, no equivalent to buffers exists naturally on the landscape. Consequently, lack of disturbance in forests immediately adjacent to riparian areas may limit some riparian-nesting species to the non-forested portion of the riparian ecotone by not providing a natural range of variation of habitats similar to post-natural disturbance (e.g., fire). As such, current management may not fulfill the habitat requirements of all riparian species. If reducing or eliminating riparian buffers does not negatively impact riparian birds, coupled with evidence that small (<100 m) buffers do not provide high quality habitat for upland nesting species (Darveau et al. 1995, Lambert and Hannon 2000, Warkentin et al. 2003), then management strategies that satisfy a larger number of species (both riparian and upland) should be implemented.

My objectives were to determine how riparian and upland bird communities change with varying amounts of harvesting around small wetlands. Short-term responses of bird species to the various harvest treatments were also investigated to assess the characteristics of early post-harvest riparian areas that influence bird community structure. Interior forest specialists (e.g., Ovenbird, *Seiurus aurocapillus*) and riparian species sometimes associated with forested wetlands (e.g., Northern Waterthrush, *Seiurus noveboracensis*) were expected to decrease in all harvest treatments but increase in larger

buffers with moderate and high retention one year post disturbance due to a "packing effect" as documented in other studies (e.g., Schmiegelow et al. 1997, Pearson and Manuwal 2001, Warkentin et al. 2003). Upland ground-nesting species and species not sensitive to edges were expected to decrease in the first year post-harvest and increase with shrub growth in subsequent years. Riparian-obligate species (e.g., Swamp Sparrow, *Melospiza georgiana*) would not be influenced by the amount of upland forest harvested whereas riparian-associated species (e.g., Common Yellowthroat, Lincoln's Sparrow, *Melospiza lincolnii*) that prefer shrubby habitats would not be affected by harvest because of the increased prevalence of shrubby habitat caused by beavers.

3.3 Methods and Study Design

3.3.1 Study Area

This study was conducted in Duck Mountain (51°39'N, 100°57'W), a forested region that straddles the Saskatchewan-Manitoba border and encompasses an area of approximately 4400km^2 (Figure 3.1). Duck Mountain is situated within the Boreal Plain near the southern-most point of this ecozone (Terrestrial Ecozones of Canada 1986). Dominant tree species in the Boreal Plain include Trembling Aspen (*Populous tremuloides*), White Spruce (*Picea glauca*), Balsam Fir (*Abies balsamea*) and Jack Pine (*Pinus banksiana*) on upland sites and Balsam Poplar (*Populous balsamifera*), White Birch (*Betula papyrifera*), Black Spruce (*Picea mariana*), and Tamarack (*Larix laricina*) on lowland (wetland) sites. Forests occur as pure or mixed stands of varying composition of these species depending on soil type, soil moisture, aspect and disturbance history (time since, type, severity; Hunter 1990). Small-scale forestry has taken place in Duck

Mountain since the early 1900s for softwood tree species and large-scale timber harvesting for timber and pulp production has taken place here since the mid-1990s. No major fires have occurred in the area since 1961 (Tardif 2004). Duck Mountain is an insular forest, with agriculture completely surrounding the mountain.

Wetlands and riparian areas cover >30% of the Boreal Plain and reach a particularly high density in Duck Mountain (National Wetlands Working Group 1997). Habitat structure and vegetation composition on the perimeter of wetlands vary greatly across the region but also commonly vary at the wetland scale. Generally, riparian areas adjacent to open-water wetlands and lakes are characterized by a gradient from the water's edge of aquatic grasses, sedges (*Carex* spp.), rushes (*Juncus* spp.) cattails (*Typha latifolia*) and lowland shrubs (*Alnus* spp., *Salix* spp.) to xeric shrubs (e.g., *Cornus canadensis*) and trees in the upland. Individual wetlands usually contain elements of various wetland types including marsh, swamp (shrubby, deciduous, conifer), fen (graminoid, shrubby, treed) or bog (graminoid, shrubby, treed; National Wetlands Working Group 1997 Locky et al. 2005). Beaver (*Castor canadensis*) harvest is prevalent >25m into the upland adjacent to most wetlands with hardwood tree species (Martell et al. 2006).

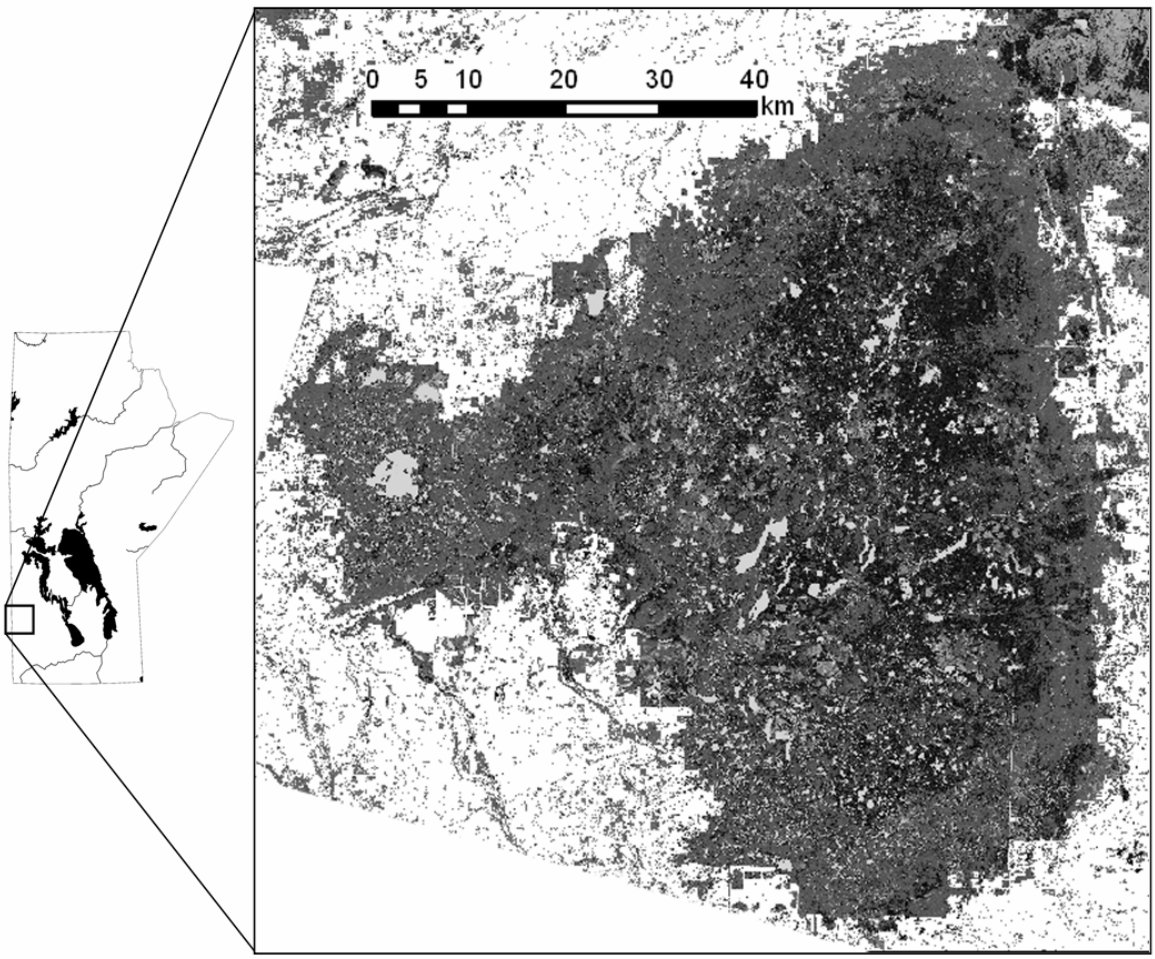

Figure 3.1. Location of the Duck Mountain study site in the province of Manitoba, Canada. Dark areas on the satellite image represent forested land, grey areas indicate large wetlands and lakes and white areas represent lands dominated by agriculture.

3.3.2 Study Design and Site Selection

Several criteria were used to select wetlands for sampling. Wetlands with adjacent hardwood (>70% *Populus* spp.) or mixedwood forests that were scheduled to be harvested the following year were selected. Wetlands were 1-35 ha in size and had a non-forested riparian zone 5-15 m wide. Selected wetlands were greater than 500 m apart from each other to ensure independent sampling. Reference sites were chosen to match vegetation attributes of experimental harvest sites. To be considered for experimental cuts, a harvest treatment had to affect at least 200 m of the forest adjacent to the wetland.

This study followed a Before-and-After Controlled Impact (BACI) design with avifaunal and vegetation surveys performed one year pre- (2004) and two years post-harvest (2005, 2006). One of three harvest treatments was randomly applied to 26 wetlands after completion of the first year of surveys; however, some treatments could not be applied as prescribed due to site characteristics (e.g., slope, conditions at time of harvest). Experimental treatments are expressed as representing approximate buffer width and percent of merchantable forest retained post-harvest in an area 50 m from the transect survey line: 1) low retention- 0 m buffer (5-35% retention; n= 7); 2) moderate retention- 10 m buffer with variable retention harvesting in the adjacent 30 m of upland (35-75% retention; n= 10; see Saskatchewan Environment 2006), and; 3) high retention- 50 m buffer (75-100% retention; n= 8). Responses of birds were assessed relative to pre-disturbance and reference sites with no anthropogenic disturbance within 500 m of the wetland (n= 9).

3.3.3 Bird Surveys

Birds were surveyed using a combined variable-width line transect and a variable-radius point count survey (Hobson and Schieck 1999). Line transects were placed parallel to the riparian-upland interface roughly corresponding to the shrub-tree or grass-shrub ecotone along the portion of the wetland where harvesting was to occur (on treatment sites). On reference sites, transects were placed where habitat types exhibited most similarity to harvest treatments. Transect length was 200 or 400 m long depending on the size of the wetland and the amount of shoreline forest to be harvested. One point count was placed on 200 m transects (at 100 m along the transect) and two point counts were placed on 400 m transects (at 50 and 350 m along the transect), 50 m upland from the water's edge. Transects were used to increase detections of 'riparian' species and point counts were used to maximize the number of upland species detected at any given site (Morissette et al. unpub. data). Transects were surveyed at an approximate rate of 10 m/minute and point counts were 10 minutes long and divided into 3 intervals (3, 2 and 5 minutes) for detection probability estimation using Farnsworth et al's (2002) removal method. Individuals detected on the transect and point count were counted as one observation in further analyses.

Surveys started at sunrise and ceased 5 hours past sunrise. They were conducted from 1- 30 June, 2004 (pre-harvest), 2005 (one year post-harvest) and 2006 (two years post-harvest) and were not conducted on days with inclement weather. Each site was surveyed once per breeding season. Field assistants were trained prior to the field season to standardize observer ability. Observers were rotated between treatments to avoid confounding observer and treatment effects.

Table 3.1. Number of 200 and 400 m transects and 10-minute point counts surveyed in four buffer treatments for birds in forest surrounding small wetlands in Duck Mountain, Manitoba 2004-2006.

Treatment	Description*	200 m	400 m	Point Counts
Reference	No disturbance within 500 m	2	7	16
Low Retention	5-35% tree retention	3	4	11
Moderate Retention	35-75% retention	6	4	14
High Retention	75-100% retention	4	4	12

* Percent retention was calculated as the proportion of area of forest left after harvest to the area of forest prior to harvest in a 50 m rectangular digital buffer around the transect survey line.

3.3.4 Vegetation Surveys

Vegetation characteristics were measured at 100 m intervals along the transect (50 m from the high water mark into the upland) and included general assessments of plant community composition and habitat structure (Table 2.2). Within each interval, visual estimates of percent canopy closure, forest stand type and dominant tree and shrub species (>20% of community) were made. Canopy height (within ±1 m) was visually estimated after calibrating estimates using a clinometer. Relative ground cover of herb, grass and moss was ranked by visual estimation. Shrub cover was visually estimated as ranks of none to low, low to moderate, moderate to high, and dense in four height categories: 0-0.5 m, 0.5-1 m, 1-3 m and 3-10 m. Assessment of wetland characteristics included emergent and riparian vegetation width and dominant plant species of each habitat. Each wetland was classified into proportions of wetland types (e.g., marsh, shrub, swamp) based on Harris et al. (2003). The amount of Downed Woody Material (DWM) was measured using a line intercept method where all downed logs greater than 10 cm in diameter were counted in two 50 m walking transects in random directions from the center of each vegetation plot. Snag density and snag decay classes were assigned at each sampling station according to Lee et al. (1997). Height, density and dominant species of trees and shrubs in regenerating cutblocks were measured in harvested sites. Vegetation surveys were conducted from mid-June to mid-August in each year of the study.

The resolution of the digital forest inventory at the riparian interface was low and did not accurately represent the composition of the non-forested portion of the riparian area. Therefore, vegetation attributes were mapped along each survey transect including the shrub line, tree line, cutblock boundary and treed residuals left post-harvest using

Geographic Positioning Systems (GPS) to modify digital forest inventories manually in Geographic Information Systems (GIS).

Digital forest inventory data were reduced to 12 habitat categories based on proportion of a polygon's area of dominant tree species to reduce the number of habitat types to fewer meaningful categories (Table 2.3). Estimates of vegetation parameters and area harvested at each sampling area were extracted using digital forest inventory data from a digital 50 m rectangular buffer around the transect survey line using GIS.

3.4 Data Analysis

3.4.1 Vegetation

A combined Principle Components Analysis (PCA) and ANOVA method was used to determine differences in pre-disturbance vegetation characteristics between treatments. Stand type proportion data extracted from digital inventories were arc-sin transformed prior to analysis. Vegetation data collected in the field were standardized to zero mean and unit variance $[(x - \bar{x})/SD]$ so that data collected on different scales (categorical versus continuous) could be directly contrasted (McCune and Grace 2002). Separate PCAs were performed for GIS, field and wetland classification data. A broken-stick method was used to determine the significance of the contribution of each principle component to the variance in the data (Peres-Neto et al. 2003). PCA and all multivariate analyses were performed using PC-Ord v4.20 (McCune and Mefford 1999), except where noted.

A simple one-way ANOVA was used to determine which vegetation variables (expressed as PCA scores) were significantly different between treatments. Non-

significant differences in pre-harvest habitat types between treatments would indicate successful selection of sites with similar vegetation characteristics. Between-treatment differences between pre-disturbance vegetation variables were of most interest, so vegetation composition of sites post-harvest were not compared. ANOVA analyses were performed using SPSS 14.0.0 (SPSS Inc., Chicago, Ill.).

3.4.2 Avian

Statistical analyses were focused on the responses of individual species, nesting (riparian, canopy, cavity, ground) and habitat (interior forest, edge-associated) guilds and bird communities (overall and riparian) to various levels of harvesting adjacent to wetlands. Samples were defined as one survey per wetland per year. Since species inhabiting the riparian zone and forest management at the riparian interface were of primary interest, only individuals detected within 50 m of the survey line or point count were included in all analyses. To account for differences in transect length, the dataset was modified so that each treatment included the same number of 200 m transects, thereby reducing the number to 26 sites (randomly selected) in all analyses except where noted.

3.4.2.1 Detection Probability Estimation

To account for biases in species detections due to disturbance parameters (i.e., disturbance type, severity of disturbance), observer aptitude, temporal effects (e.g., time, date) and habitat types (White 2005), detection probabilities were estimated for species (>80 detections) and groups of species (where individual species' counts <80; Appendix

A1) on point counts using Farnsworth et al's (2002) removal method. Huggins' (1989) closed-capture models were run in program RMark (R Development Core Team 2007; Laake 2007) with an interface to program MARK (White 2006). Each point count was considered a separate 'capture' event used in detection probability estimation. An effort parameter expressed as the natural log of each time interval was added to each model to account for unequal likelihood of detecting new individuals in successive time intervals. Akaike's (1973) Information Criterion (AICc; adjusted for small sample size) was used to select the most parsimonious models. All parameters in the top models were averaged to derive beta estimates for each parameter (Burnham and Anderson 2002) and then applied back to the data to derive individual-specific detection probability estimates. Since transects were placed parallel to the riparian habitat, detection probabilities were not estimated for species detected only on transect because this method of data collection violates the assumptions of DISTANCE sampling (Buckland et al. 2001). This is the only program currently available to estimate detection probabilities on line transects without multiple observers or within-year visits. Therefore, data used in analyses were adjusted for detectability on point counts but not for transects.

3.4.2.2 Change in Abundance

To determine the responses of species and nesting guilds to treatments between years, I used repeated measures Analysis of Variance (ANOVA). Prior to analysis, non-normally distributed count data were log transformed [ln(x+ 1)] to meet the assumptions of univariate analyses (Gotelli and Ellison 2004). Repeated measures ANOVA was performed on 22 species and six guilds detected greater than 30 times within 50 m of the

survey line. Year and treatment were included as within-subject variables and between subject factors in the analysis, respectively.

To ensure equality of the variances of the differences between levels of the repeated measures factor (sphericity), I used Mauchly's test using a significance level of $p<0.01$. Non-significant results from this test would indicate that this assumption was met (Gotelli and Ellison 2004). Significant year*treatment (*Y* **T*) effects from the repeated measures ANOVA would indicate a significant change in abundance in at least one treatment relative to reference sites or from pre-harvest levels. Graphs of riparian species with non-significant effects from this analysis were included to show general responses of species to different levels of harvest.

3.4.2.3 Bird Community Response to Harvest

Multi-response permutation procedures (MRPP) with the relative Euclidean distance measure among centroids were employed to assess the differences in the overall and riparian bird community composition between treatments and years. MRPP is a non-parametric procedure used to test the null hypothesis of no difference between *a priori* groups of data. Between-group differences are measured using a homogeneity test (A), where a value <0 indicates significantly greater homogeneity compared to random (McCune and Grace 2002). MRPP was favored over parametric tests of community response (e.g., Multivariate Analysis of Variance- MANOVA) because the dataset consisted of many zeroes and many species were rare. Furthermore, assumptions of multivariate normality and homogeneity of variances are not required with this method (McCune and Grace 2002). Within-group differences (across years) would suggest bird

communities changed in response to level of harvest. For multivariate analyses, species detected less than three times were removed from the data.

To assess the environmental variables responsible for shaping bird communities, Multivariate Regression Trees (MRTs) were produced using the Bray-Curtis distance measure for the overall and, separately, for the riparian bird communities for the two years immediately post-harvest (2005 and 2006). Graphical representations of MRTs illustrate bird community response to various environmental gradients and can also be used to predict species distributions (De'ath 2002). Splits (nodes) on the tree are chosen based on a dissimilarity matrix of sum-of-square distances (SSD) between sites (De'ath 2002). For MRT analyses, all observations were standardized to the number of birds detected per unit length of transect to account for varying transect lengths. MRTs were run using the R library (R Development Core Team 2007), MVPART (Therneau and Atkinson 2005)

Species abundances were included as response variables and proportional habitat data derived from GIS and ground vegetation variables were the explanatory variables. Forty-one and 39 species were entered into the overall MRT analysis for 2005 and 2006, respectively. Ten species were used in the riparian analysis for both years post-harvest. Multiple trees for each data set were produced with the final tree selected based on the cross-validation error (CVE) and the tree within one standard error (SE) of the best tree. Low CVE values signify greater predictive accuracy (Breiman et al. 1984).

Indicator species analysis (ISA; Dufrene and Legendre 1997) was used to determine species characterizations of each leaf (group) in the tree (using $p<0.1$). Indicator Species Analysis uses relative frequency and relative abundance of species within groups selected

a priori relative to a randomized data matrix to assess indicator values for a species (Dufrene and Legendre 1997), where an indicator value of 0 signifies a poor indicator and 100 a perfect indicator.

3.5 Results

3.5.1 Vegetation

The first three axes of the pre-harvest GIS PCA explained 92.9% of the variance in the data, and represented area of hardwood (Axis 1- 53.6%) and area of shrub (Axis 2- 22.0% and Axis 3- 17.3%). Eigenvalues suggested that three axes adequately represented the variability in pre-disturbance ground vegetation data. Axis 1 was associated with amount of herbaceous cover (23.1% of the variance), Axis 2 with average shrub height 0-0.5m (16.8%) and Axis 3 with downed woody material (13.7%) accounting for a total of 53.6% of the variance in the pre-disturbance ground vegetation data. One axis, associated with proportion of meadow marsh, most appropriately represented the variability (80.03%) in the wetland classification data as indicated by the broken-stick eigenvalue. Results from one-way ANOVA showed that principle components of the pre-harvest vegetation variables were not statistically different between treatments (all *p's*>0.14).

3.5.2 Avian

Eighty-one species were detected within 50 m of the observer at 107 sampling stations representing 2381 individual birds across all years. The three species detected most often on surveys were Chestnut-sided Warbler (*Dendroica pensylvanica*, 9% of observations), White-throated sparrow (*Zonotrichia albicollis*, 7%) and American

Redstart (*Setophaga ruticilla,* 6%). The most abundant riparian species were Swamp Sparrow (*Melospiza georgiana,* 5%), Common Yellowthroat (*Geothlypis trichas,* 5%) and Song Sparrow (*Melospiza melodia,* 4%), accounting for 14% of total number of individuals detected.

3.5.2.1 Species and Guild Response to Level of Harvest

Species and guilds showed variable responses to amount of harvest. All species and guilds met the assumptions of sphericity (Mauchly's test; $p>0.01$) for univariate repeated measures ANOVA. Eight of 22 species and two of six guilds differed in abundance with year, treatment or year*treatment effects ($p<0.1$; Table 3.1). Abundance of Cedar Waxwing (*Bombycilla cedrorum,* $F_{3,22}= 2.67$, $p<0.1$), Chestnut-sided Warbler ($F_{3,22}= 3.21$, $p<0.05$), Song Sparrow ($F_{3,22}= 4.14$, $p<0.05$) and the riparian guild ($F_{3,22}= 5.70$, $p<0.005$) differed between treatments (Figures 3.2 and 3.3). Cedar Waxwing abundance did not change in any treatments except reference sites where they exhibited a decrease and then a sharp increase in the two years post-harvest, respectively. Increases in abundance in reference and high retention and no change in abundance in low and moderate retention were evident for the Chestnut-sided Warbler. Song Sparrow, a riparian/open habitat generalist, increased in abundance in harvested sites but not on reference sites. Abundance of species in the riparian guild increased in harvest with low and high retention and remained stable with moderate harvest and reference sites.

Within-treatment (between-year) differences in abundance were observed for American Redstart ($F_{1,22}= 6.52$, $p<0.05$), Chestnut-sided Warbler ($F_{1,22}= 3.98$, $p<0.1$), Myrtle Warbler (*Dendroica coronata,* $F_{1,22}= 3.88$, $p<0.1$), Song Sparrow ($F_{1,22}= 9.14$,

$p<0.01$) and the interior nesting species guild ($F_{1,22}= 7.58$, $p<0.05$). American Redstarts increased significantly in moderate and high retention sites with no change in abundance witnessed for reference sites. This species exhibited a decrease in abundance the first year after harvest and then a moderate increase in abundance the second year post-harvest. Myrtle Warblers showed slight decreases in abundance from pre-disturbance levels in all treatments except for sites with high retention where abundance appeared stable. Interior forest nesting species showed a significant decrease in sites with low tree retention with smaller decreases in moderate retention sites. Species in this guild did not change in abundance in reference sites and in sites with high retention.

Common Yellowthroat ($F_{3,22}= 4.47$, $p<0.05$), Mourning Warbler (*Oporornis philidelphia,* $F_{3,22}= 6.48$, $p<0.005$), White-throated Sparrow ($F_{3,22}= 2.48$, $p<0.1$) and interior forest nesting species ($F_{3,22}= 3.48$, $p<0.05$) exhibited significant year*treatment interactions. Common Yellowthroat, a riparian-associated species, showed variable responses to amount of harvest with overall increases in abundance with high retention and no harvest and overall decreases with moderate retention after harvest. Mourning Warblers showed a sharp increase in abundance with low retention and decreases in abundance with moderate retention and reference sites. Variable responses to harvest were observed with the White-throated Sparrow where an increase in abundance was seen in high retention treatments in both years post-harvest and a decrease in the second year after harvest in reference sites and no change in abundance in low and moderate treatments. Changes in abundance for riparian nesting species including Northern Waterthrush (*Seiurus noveboracensis*), Swamp Sparrow and Yellow Warbler (*Dendroica petechia*; all p's>0.1) are shown to illustrate general trends for these species (Figure 3.4).

Table 3.2. Results from repeated measures AVOVA tests contrasting abundance of birds detected >30 times at sampling stations in riparian habitats one year prior to and two years after harvest, at unharvested reference sites (n= 9) and three harvesting treatments: 1) low (5-35%; n=6), 2) moderate (35-75%; n=6), and, 3) high (75-100%; n=6) buffer area retention in Duck Mountain, MB in 2004-2006. Latin names for species are listed in Appendix B1.

	Year			Treatment			Year*Treatment		
Species	**df****	**F**	**p**	**df**	**F**	**P**	**df**	**F**	**p**
American Redstart	1,22	6.516	0.019	3,22	1.771	0.184	1,22	0.333	0.802
American Robin	1,22	0.061	0.807	3,22	1.094	0.374	1,22	0.830	0.492
Black-and-white Warbler	1,22	0.326	0.574	3,22	0.735	0.543	1,22	0.538	0.662
Brown-headed Cowbird	1,22	0.226	0.639	3,22	1.850	0.169	1,22	0.940	0.439
Blackburnian Warbler	1,22	0.471	0.500	3,22	1.848	0.169	1,22	0.621	0.609
Cedar Waxwing	1,22	1.702	0.206	3,22	2.672	0.074	1,22	0.460	0.713
Chipping Sparrow	1,22	0.001	0.976	3,22	0.714	0.554	1,22	0.588	0.629
Common Yellowthroat*	1,22	0.986	0.332	3,22	1.856	0.168	1,22	4.474	0.014
Chestnut-sided Warbler	1,22	3.979	0.059	3,22	3.206	0.044	1,22	1.384	0.275
Least Flycatcher	1,22	0.057	0.814	3,22	1.447	0.258	1,22	1.352	0.284
Mourning Warbler	1,22	0.099	0.757	3,22	1.404	0.269	1,22	6.482	0.003
Myrtle Warbler	1,22	3.883	0.062	3,22	0.357	0.784	1,22	0.550	0.654
Northern Waterthrush*	1,22	0.294	0.593	3,22	0.285	0.836	1,22	0.968	0.426
Ovenbird	1,22	0.827	0.374	3,22	0.489	0.694	1,22	1.069	0.384
Rose-breasted Grosbeak	1,22	0.000	0.988	3,22	0.368	0.777	1,22	1.393	0.273
Ruby-crowned Kinglet	1,22	1.935	0.179	3,22	0.812	0.501	1,22	0.815	0.500
Red-eyed Vireo	1,22	0.098	0.757	3,22	0.904	0.456	1,22	0.802	0.507
Song Sparrow*	1,22	9.138	0.006	3,22	4.142	0.019	1,22	0.985	0.419
Swamp Sparrow*	1,22	1.228	0.280	3,22	0.967	0.427	1,22	2.042	0.139
White-throated Sparrow	1,22	0.387	0.540	3,22	0.702	0.561	1,22	2.476	0.089
Yellow Warbler*	1,22	2.612	0.121	3,22	0.856	0.479	1,22	0.272	0.845
Guild									
Riparian	1,22	2.716	0.114	3,22	5.699	0.005	1,22	0.893	0.461
Interior Forest	1,22	7.579	0.012	3,22	2.189	0.119	1,22	3.479	0.034
Edge-associated	1,22	2.15	0.157	3,22	0.678	0.575	1,22	1.069	0.383
Canopy	1,22	0.240	0.629	3,22	0.610	0.616	1,22	1.848	0.170
Cavity	1,22	0.084	0.775	3,22	0.124	0.945	1,22	0.240	0.867
Ground	1,22	1.669	0.210	3,22	0.893	0.461	1,22	1.612	0.217

*Riparian species

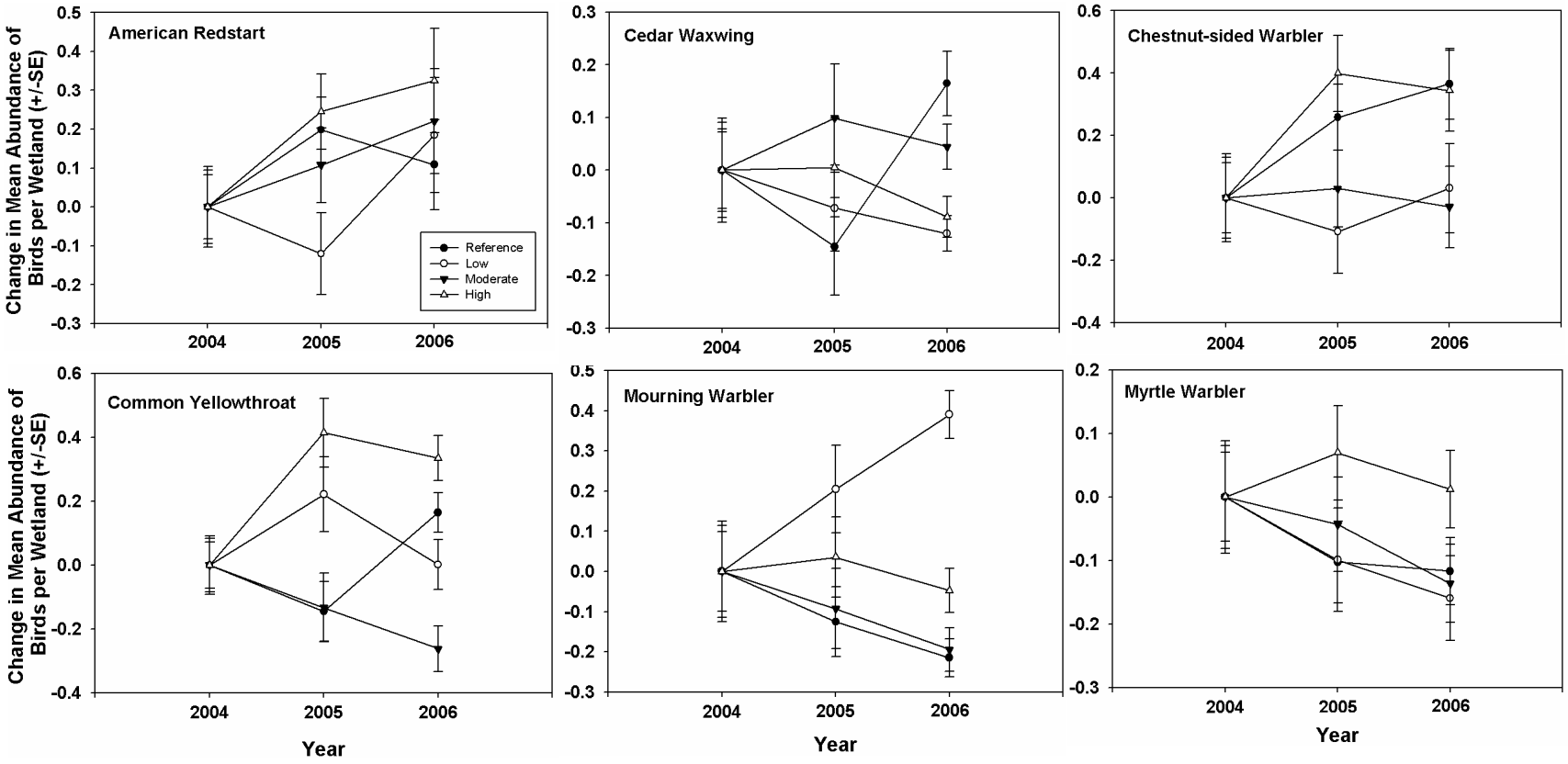

Figure 3.2. Change in mean abundance (±1SE) per sample station of American Redstart, Cedar Waxwing, Chestnut-sided Warbler, Common Yellowthroat, Mourning Warbler and Myrtle Warbler in unharvested reference sites (n= 9) and three levels of harvest around wetlands: 1) low (5-35%; n= 6), 2) moderate (35-75%; n= 6), and, 3) high (75-100%; n= 6) retention one year prior to harvest (2004) and two years after harvest (2005, 2006) in Duck Mountain, MB. Latin names for species are given in Appendix B1.

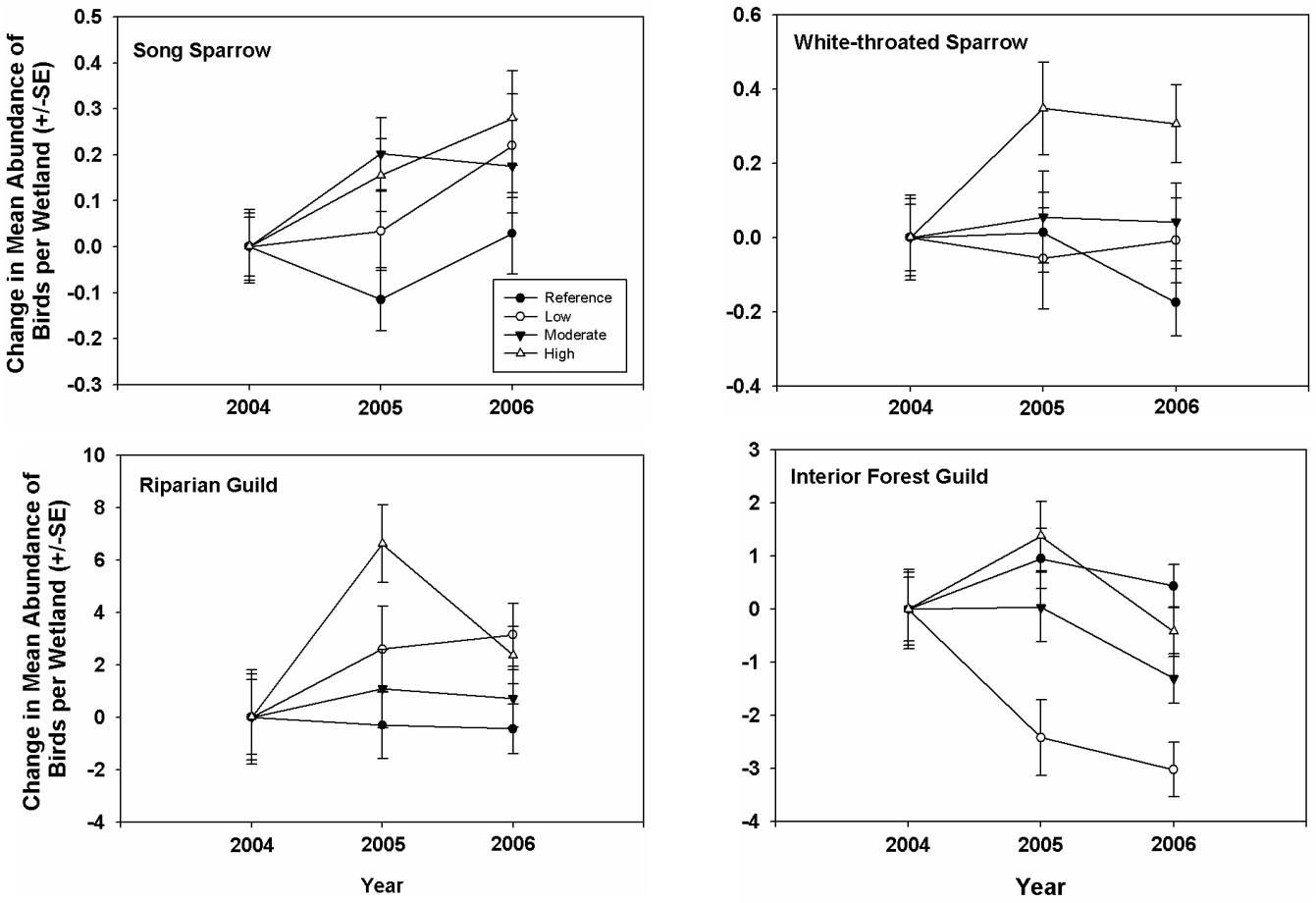

Figure 3.3. Change in mean abundance (±1SE) per sample station of Song Sparrow, White-throated Sparrow and riparian and interior forest nesting species in unharvested reference sites (n= 9) and three levels of harvested shoreline forests around wetlands: 1) low (5-35%; n= 6), 2) moderate (35-75%; n=6), and, 3) high (75-100%; n=6) retention one year prior to harvest (2004) and two years after harvest (2005, 2006) in Duck Mountain, MB. Latin names for species are given in Appendix B1.

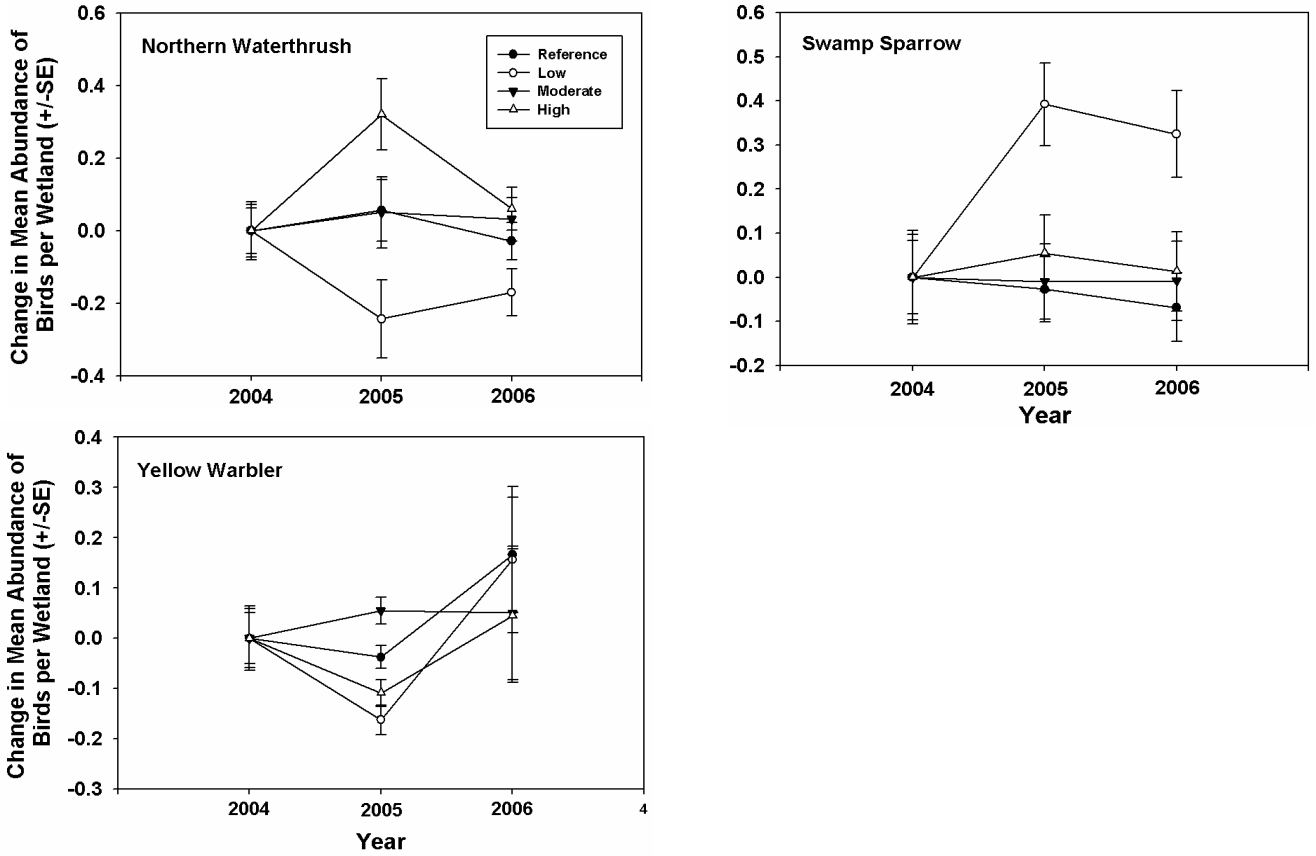

Figure 3.4. Change in mean abundance (±1SE) per sample station of Northern Waterthrush, Swamp Sparrow and Yellow Warbler in unharvested reference sites (n= 9) and three levels of harvested shoreline forests around wetlands g: 1) low (5-35%; n= 6), 2) moderate (35-75%; n= 6), and, 3) high (75-100%; n=6) retention one year prior to harvest (2004) and two years after harvest (2004, 2005) in Duck Mountain, MB. All interactions were non-significant (Repeated measures ANOVA, *p's*>0.1). Latin names for species are given in Appendix B1.

3.5.2.2 Bird Community Response to Harvest

Results from MRPP indicated that differences in overall bird community composition existed between years and treatments ($p<0.05$; Table 3.3). Non-significant differences between treatments in the overall pre-disturbance bird community ($A= -0.0139$, $p= 0.92$) were contrasted by more heterogeneous communities one year post-harvest ($A= 0.014$, $p<0.05$) and significant differences two years after harvest ($A= 0.0243$, $p<0.005$). Within-treatment differences for all species across years were seen in all harvest treatments (low retention: $A= 0.0215$, $p<0.05$, moderate retention: $A= 0.514$, $p<0.005$, high retention: $A= 0.0363$, $p<0.005$) but not on reference sites ($A= 0.004$, $p= 0.32$). The riparian community did not exhibit any change with year (all *p's*>0.2) or across treatments (all *p's*>0.7) using MRPP.

Table 3.3. Chance-corrected within group agreement (A) and significance values (p) of Multi-response Permutation Procedures (MRPP), testing for differences between treatments and years and within treatments (across years). Tests were performed for the overall and riparian bird communities one year prior to harvest (2004) versus two years after harvest (2005, 2006) for unharvested reference sites (n= 9) and three levels of harvesting: 1) low (5-35%; n= 7), 2) moderate (35-75%; n= 10), and, 3) high (75-100%; n= 8) retention in Duck Mountain, MB.

Contrast	**Year**					
Year (between-subject)	**2004**		**2005**		**2006**	
	A	p	A	p	A	p
All Species	-0.014	0.924	0.014	0.021	0.024	0.005
Riparian	-0.034	0.941	0.007	0.319	0.010	0.243

Treatment (within-subject)*	**All Species**		**Riparian**	
	A	p	A	p
Reference	-0.014	0.881	-0.024	0.782
Low Retention	0.022	0.011	-0.047	0.999
Moderate Retention	0.051	0.000	-0.018	0.708
High Retention	0.036	0.004	-0.062	0.987

* Percent retention was calculated as the proportion of area of forest left after harvest relative to the area of forest prior to harvest in a 50 m square digital buffer around the survey line.

The multivariate regression trees for the entire bird community one (2005) and two (2006) years after harvest resulted in two splits and three terminal nodes (Figure 3.5a and b). In 2005, the tree split first with amount of hardwood/mixedwood forest (28.5%). Amount of hardwood/mixedwood was further split into terminal nodes based on amount of harvest (41.1%). Indicator species of the first split (Node 3) associated with hardwood/ mixedwood greater than 28.5% included Brown Creeper (*Certhia americana*), Black-and-white Warbler (*Mniotilta varia*), Red-breasted Nuthatch (*Sitta canadensis*) and White-throated Sparrow (Table 3.4). At the second split (Node 1) species associated with amount of harvest greater than 41.1% included American Redstart and Chipping Sparrow (*Spizella passerina*) while no species were indicative of amount of harvest less than this amount. The selected environmental variables accounted for 7.7% of the variance in the data.

The MRT for 2006 split twice based on amount of forest harvested, once at 33.0% and again at 16.1%. The environmental variables used in the 2006 MRT accounted for 21.3% of the variance in the data. Indicator species of the first split (Node 1) with amount of forest harvest >33.0% were American Redstart, Blackburnian Warbler (*Dendroica fusca*), Boreal Chickadee (*Poecile hudsonicus*), Chestnut-sided Warbler (*Dendroica pensylvanica*), Rose-breasted Grosbeak (*Pheucticus ludovicianus*) and White-throated Sparrow. Chipping Sparrow and Ruby-crowned Kinglet (*Regulus calendula*) were indicator species of the second split with >16.1% forest harvested. The Mourning Warbler was the only species indicative of area of forest harvested <16.1% (Node 2). The MRTs for the riparian species in 2005 split once based on proportion of shrub (> or < 21.9%). Proportion of graminoid (13.0%) and shrub (15.4%) produced the first

and second splits in the 2006 riparian community data. Alder Flycatcher and Lincoln's Sparrow were indicator species of the first terminal node (Node 3) associated with graminoid vegetation <13.0%. Only one species, the Swamp Sparrow, was indicative of the second split in the data and was associated with proportion of shrub <15.4% (Node 1). The amount of variance explained by the environmental variables in the 2005 and 2006 riparian MRTs were 18.2% and 28.6%, respectively. High cross-validation errors (CVE) were associated with each of the MRTs (2005: all- CVE= 1.17, riparian- CVE= 1.21; 2006: all- CVE= 1.25, riparian- CVE= 1.21).

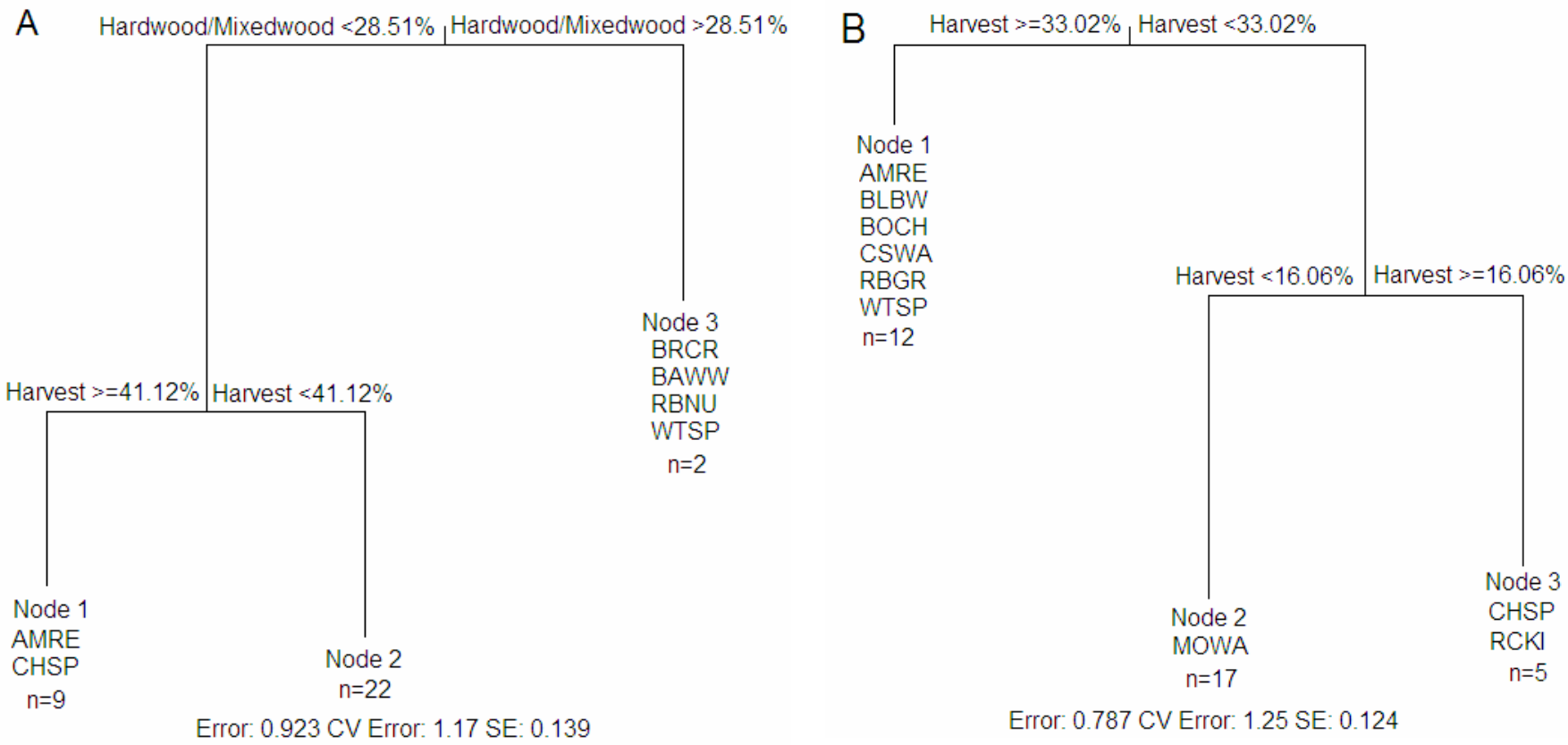

Figure 3.5. Multivariate regression trees and their associated errors, cross-validations (CV) errors and standard errors (SE) for overall bird communities one (a) and two (b) years post-harvest in Duck Mountain, MB. Codes for indicator species from Indicator Species Analysis and the number of sites (n) associated with each node (See Table 3.4 for species names and indicator values for each MRT).

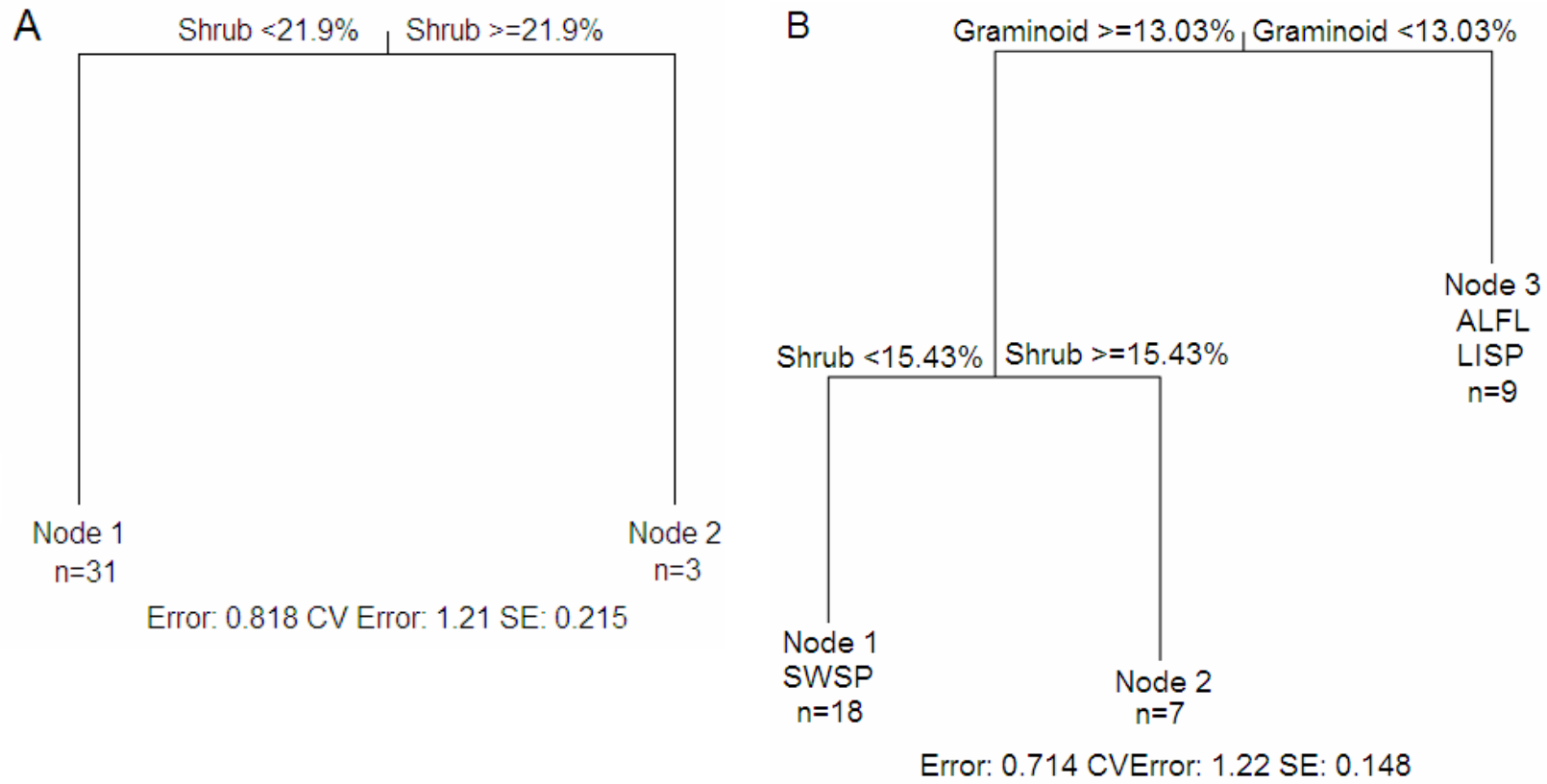

Figure 3.6. Multivariate regression trees and their associated errors, cross-validations (CV) errors and standard errors (SE) for riparian bird communities one (a) and two (b) years after harvest in Duck Mountain, MB. Codes for indicator species derived from indicator species analysis and the number of sites (n) associated with each node (See Table 3.4 for species names and indicator values for each MRT).

Table 3.4. Species with significant indicator values derived from indicator species analysis for terminal nodes of each multivariate regression tree, their associated species codes, and significance values for overall and riparian bird communities one (2005) and two (2006) years after harvest. Latin names for species are given in Appendix B1.

2005	Node	Species Code	Species	Indicator Value	p-value
Overall Community	1	AMRE	American Redstart	55.6	0.053
	1	CHSP	Chipping Sparrow	54.6	0.021
	2	BRCR	Brown Creeper	46.5	0.063
	2	BAWW	Black-and-White Warbler	88.4	0.009
	2	RBNU	Red-breasted Nuthatch	45.7	0.063
	2	WTSP	White-throated Sparrow	77.4	0.065
Riparian Community	n/a	n/a	n/a	n/a	n/a
2006					
Overall Community	1	AMRE	American Redstart	43.3	0.075
	1	BLBW	Blackburnian Warbler	32.4	0.034
	1	BOCH	Boreal Chickadee	30.2	0.076
	1	CSWA	Chestnut-sided Warbler	61.4	0.007
	1	RBGR	Rose-breasted Grosbeak	32.5	0.048
	1	WTSP	White-throated Sparrow	49.6	0.059
	2	MOWA	Mourning Warbler	53.8	0.076
	3	CHSP	Chipping Sparrow	79.4	0.002
	3	RCKI	Ruby-crowned Kinglet	35.2	0.079
Riparian Community	1	SWSP	Swamp Sparrow	57.3	0.045
	2	n/a	n/a	n/a	n/a
	3	ALFL	Alder Flycatcher	26.1	0.073
	3	LISP	Lincoln's Sparrow	49.6	0.003

3.6 Discussion

This study provides initial insight into the responses of bird communities inhabiting the upland-riparian interface of small boreal wetlands to various levels of forest harvesting. Previous research has focused primarily on assessing the value of treed riparian buffer strips to upland-nesting birds and the inclusion of species primarily using the non-forested riparian area is unique. While forest harvesting may not directly impact riparian habitats, birds inhabiting riparian areas may be indirectly affected as harvesting may produce or modify habitats suitable for some riparian species, change predator community dynamics (Bayne and Hobson 1998, Hannon et al. 2002) or change abiotic parameters associated with wetlands or riparian areas (e.g., water level; Van Damme et al. 2003). As well, data from this study are important for understanding riparian bird community interaction with upland habitat and in offering alternatives to current boreal forest riparian management.

Since the shoreline forests, riparian areas and wetland vegetation parameters at sampling sites did not differ in pre-disturbance vegetation composition and structure, I was able to compare responses of bird communities across treatments and years. Species inhabiting the riparian-upland interface of small wetlands exhibited variable responses to level of harvest. Only five riparian species were detected enough times to warrant the use of repeated measures ANOVA to test responses. Therefore, riparian species were also grouped into a single guild for analysis. Variable responses of the Common Yellowthroat made it difficult to directly interpret a response to amount of harvest; however, increased abundance in high retention and reference sites over two years suggests that this species prefers habitats that have more canopy and shrub structure and avoid recently harvested

sites. Song sparrows showed increases in abundance in all harvest treatments relative to reference sites where no increase was observed. As a habitat generalist, this species likely benefits from open habitats created by harvest (Arcese et al. 2002).

Three riparian species, the Northern Waterthrush, Swamp Sparrow and Yellow Warbler showed non-significant year or treatment responses in this study. However, trends in their abundances are relevant to forest managers and are discussed here. Similar to Warkentin et al.'s (2003) study, Northern Waterthrushes exhibited a slight 'packing' effect in the largest buffers (high retention) and an avoidance of sites with the smallest buffers (low retention) one year after harvest. Although Swamp Sparrows showed no statistically significant changes in abundance to amount of harvest, they appear to have increased considerably in sites with low levels of retention. Sites with low tree retention may support a reduced predator community or they may enable Swamp Sparrows to better detect predators.

As I predicted, riparian bird community composition did not change with year after harvest or treatment relative to pre-disturbance composition. Some species typically thought to be riparian-associated, including the Common Yellowthroat and Lincoln's Sparrow, were detected in regenerating cutblocks adjacent to wetlands in the second year post-harvest. However, no substantial increases in abundance of riparian birds with increasing amounts of harvest were detected in the first two years post-harvest except for Song Sparrow. Regenerating aspen trees were fairly short (<2m) in the second year after harvest and may not closely mimic riparian habitat well. These species may become more abundant three to five years after harvest as aspen suckers grow taller and are recognized as a surrogate for riparian habitat (Twedt et al. 1999, Schieck and Song 2006).

Open habitat species including White-throated Sparrow increased in all treatments relative to pre-treatment abundance and reference sites. Two shrub-associated species, American Redstart and Chestnut-sided Warbler, exhibited declines in abundance only in the first year post-harvest in sites with the lowest amount of retention indicating that eliminating buffers removes suitable habitat for these species in the first year post-harvest. The Mourning Warbler (*Oporornis philidelphia*) increased in abundance in low retention sites and declined in all other treatments suggesting that there may be competition between American Redstart and Chestnut-sided Warbler for higher quality habitat (i.e., competitive exclusion; Hardin 1960). The high amount of shrub habitat present between riparian areas and shoreline forest (resulting from beaver harvest; Martell et al. 2006) in the Duck Mountains may 'force' more individuals from harvested areas here and provide habitat as forests regenerate. High proportion of shrub was present even on sites with high amounts of forestry because forestry operations did not disturb (e.g., trample) the shrubby areas. These species are known to inhabit regenerating cutblocks 5-10 years post-harvest and, as such, likely benefit from increased harvest in shoreline forest (Hobson and Bayne 2000a). Interpretations of changes in abundance for Cedar Waxwing are complicated by the fact that this species is not generally territorial, breeds later in the season corresponding to peak periods in fruit abundance and tends to breed where fruit crops are locally abundant (Witmer et al. 1997).

In agreement with other studies (e.g., Hannon et al. 2002, Hanowski et al. 2005, Shirley and Smith 2005), interior forest nesting species (or those with larger territory sizes) responded negatively to increasing amounts of harvest. Abundance of interior forest nesting species declined in all harvest treatments over the three years of the study

with the greatest decrease in sites with low retention. Species in this guild increased in the high retention treatment, appeared stable in the moderate retention treatment in the first year after harvest and showed a delayed negative response in all treatments the second year after harvest. Contrary to other studies (Lambert and Hannon 2002), abundance of Ovenbirds (*Seiurus aurocapillus*) was not related to harvest level or year. This was unexpected because this species prefers contiguous habitats (Bayne and Hobson 2002). However, when their response to harvest at a greater spatial scale (100 m from the survey line) was examined, Ovenbird abundance decreased, implying that scale of the chosen analysis for some species may not be adequate. Similar to other studies (Darveau et al. 1995, Schmiegelow et al. 1997, Hanowski et al. 2002), the overall bird community exhibited a lag response to harvest with greater differences in community composition evident two years after harvest. As such, a delayed response to harvest treatment by some species may not be captured with the three-year time scale of this study.

The MRTs used in this analysis were useful in describing the environmental variables responsible for shaping bird community composition at the riparian interface of small wetlands. Low amounts of variation (7.7%) explained by the environmental variables of the overall bird community to amount of hardwood and mixedwood forest in the first year after harvest gives more support for a delayed response by some species to harvest. This suggests that factors other than amount of forest retained at the riparian interface (e.g., cutblock size, retention in the cutblock, shrub cover created by beaver) may be influencing bird response. The MRT for the second year after harvest show that the amount of harvest was the dominant environmental parameter shaping bird communities and also had a greater proportion of the variance explained in this MRT (18.6%). Species

associated with high amounts of harvest (>33%) including American Redstart and Chestnut-sided warbler may be crowding into shrubby habitats created by beavers at the riparian edge as the upland forest regenerates as previously mentioned. Other species associated with higher amounts of harvest include habitat generalists (White-throated sparrow and Rose-breasted Grosbeak) and a resident species (Boreal Chickadee). I classified the Blackburnian Warbler as an interior species, however, it also was an indicator for high proportion of harvest. In years of Spruce Budworm (*Choristoneura fumiferana*) outbreak, this species may be pushed out of primary habitats by other species (e.g., Cape May Warbler, Canada Warbler; Patten and Burger 1998) and this may be why it is inhabiting sites with high amounts of harvest.

Splits associated with non-merchantable habitats (graminoid and shrub) in the post-harvest riparian bird community data and few indicator species for each split supports other multivariate analyses (i.e., MRPP) that suggest that level of harvesting adjacent to boreal riparian habitats may not affect abundance and composition of riparian bird communities. Indicator species for the MRTs were only present in the second year post-harvest where Swamp Sparrows were associated with sites with <15.4% shrub cover. This species prefers wetland habitats with large areas of graminoid vegetation (Mowbray 1998), so this result is not surprising. Similarly, results that show Alder Flycatchers and Lincoln's Sparrows associated with sites with <13.0% graminoid vegetation are consistent with their habitat preferences. Large cross-validation error (CVE) values (1.17-1.25) associated with the resulting trees indicate poor predictive power for other datasets and indicated high variability within this dataset (De'ath 2002).

Reference sites used in BACI designs should be used to show temporal variation in

species' abundance so that responses of birds to treatments can be accurately assessed (Zar 2005). Abundance of birds in the chosen reference sites were generally consistent across years with the exception of a few species (e.g., Chestnut-sided Warbler). Inter-annual variation in species' owing to increased amounts of shrubby vegetation counts may have eliminated or reduced the significance of year or treatment effects on harvested sites. More years of pre-harvest data would have been optimal to assess longer-term variation in bird abundance (Hannon and Schmiegelow 2002) in the absence of disturbance.

Variability in responses by different species to level of harvest might be a product of the high amount of residual forest left after harvest in the adjacent cutblock, the high prevalence of beaver activity (e.g., shrubby habitat), high variability in cross-seasonal population dynamics (Maron et al. 2005), competitive exclusion or the low number of replicates and detections. For example, when GIS data were analyzed within 100 m of the survey line, no harvested site had >63% of the area harvested. Furthermore, the first year of surveys (2004) was unusually cold (e.g., snowfall on 22 June) and this may have resulted in reduced prey abundance and lower breeding success (Rotenberry and Wiens 1991, Heltzel and Earnst 2006) causing changes in populations in post-harvest years. However, abundance measurements should have reflected these trends in all of the treatments. Low sample sizes in each of the harvest treatments (n= 6) for some analyses give low power to detect trends in the dataset. More samples and more surveys in pre- and post-harvest years would provide a better representation of riparian bird community dynamics in response to upland forest harvesting relative to known bird community variation across years.

Many riparian species (Common Grackle *Quiscalus quiscalus*, Common Snipe *Gallinago delicata*, Lincoln's Sparrow *Melospiza lincolnii*) were detected too few (<30) times to warrant the use of repeated measures ANOVA. Furthermore, many riparian/wetland species were rare (detected <10 times in three years) including Greater Yellowlegs (*Tringa melanoleuca*), Rusty Blackbird (Euphagus carolinus), and Olive-sided Flycatcher (*Contopus cooperi*). These species may not be surveyed well using point count techniques because they typically have large home ranges and may not prefer the habitat types I sampled (i.e., merchantable timber). However, they may be indirectly impacted by harvest through changes in water levels (Lariviere and Lepage 2000, Van Damme et al. 2003), changes in predator dynamics (Ibarzabal and Desrochers 2004) or by changes in landscape structure and composition as seen by waterbird species (e.g., Connor and Gabor 2006). Directed studies should investigate how these species are impacted by forestry, particularly at the landscape scale.

3.7 Management Implications

Previous research has established that smaller (<100 m) buffers have low habitat suitability for upland species (e.g., Hannon et al. 2002, Hanowski et al. 2005). In this study, two riparian species showed increases in abundance with high levels of harvesting around small wetlands one and two years after harvest. Three riparian species showed no response to amount of harvest relative to pre-disturbance and to reference sites. Riparian bird communities (10 species) showed no response to amount of harvest in the two years after harvest. High amounts of shrub habitat from beaver harvest adjacent to wetlands may also dampen the effect of forest harvesting on riparian species in the sites studied.

Furthermore, I was only able to directly test the responses of five riparian species, most of which are habitat generalists known to benefit from harvesting (e.g., Song Sparrow), to amount of shoreline forest harvest. Other species with limited distributions or more specific habitat requirements may be most vulnerable to forestry and future studies should identify their population limitations due to forestry. Many other riparian species were detected too few times including many species (e.g., Olive-sided Flycatcher *Contopus cooperi*, Yellow-bellied Flycatcher *Empidonax flaviventris*) that may not benefit from harvesting. These species and those considered 'at risk' (e.g., Rusty Blackbird) should be considered in harvesting scenarios, especially in areas where they are known to breed. I assume that the regeneration of cutblocks adjacent to riparian habitats in subsequent years (3-10 years) after harvest would provide habitat for some riparian species (e.g., Common yellowthroat, Lincoln's Sparrow) and the elimination of buffers may benefit these riparian species.

Therefore, current management guidelines that prescribe strict buffer widths adjacent to water bodies across the landscape should be reviewed to determine their effectiveness in providing habitat for the full suite of riparian and upland species. To address conservation issues of all boreal species that may be impacted by forest harvesting, riparian management guidelines should be coupled with landscape management approaches to account for the amount of shoreline forest being retained on the landscape. With a decreasing amount of old-growth forest being left on the landscape, riparian buffer strips may contain the only remaining old-growth forest. Results that show lack of a negative response to harvest by riparian birds allows for the exploration of management scenarios that strategically place post-harvest residuals on the landscape, as suggested by

Devito et al. (2000) and Hannon et al. (2002). Such management may involve aggregating large residual patches from linear buffer strips to larger core areas but requires further testing before being implemented.

In addition to providing poor quality habitat, narrow treed buffer strips are prone to increased blowdown (Ruel et al. 2000), beaver harvest (Martell et al. 2006) and missing succession to early seral stages (Lee and Barker 2005). Long-term consequences of applying broad buffer width prescriptions across the landscape with little variation are unknown but may have negative implications for timber supply and habitat suitability (Lee and Barker 2005). Therefore, it is important for forest managers to consider management scenarios that incorporate shoreline forests into forest harvesting schemes (e.g., cutting to the edge of the high water mark).

The complete elimination of riparian buffers on the landscape is not advocated because buffers serve other important functions (e.g., organic inputs, downed trees for fish habitat, aesthetics, maintaining integrity of riparian structure). Additionally, there may be a disproportionate benefit to maintaining larger tracts of forest next to water bodies due to increased insect prey abundance (Whitaker et al. 2000, Nakano and Murakami 2001) and greater habitat heterogeneity (Harper and Macdonald 2001, Macdonald et al. 2006). This may increase nest success and provide habitat for a greater number of species while providing other benefits for birds and other wildlife (e.g. migration corridors; Machtans et al. 1996, Mosley et al. 2006).

3.8 Literature Cited

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Appendix B1. Common name, scientific nomenclature, four-letter identifier code and abundance of species detected in Duck Mountain, MB, in an experimental study contrasting species responses in undisturbed reference sites to three harvest treatments: 1) low (5-35%), 2) moderate (35-75%), and, 3) high (75-100%) retention one year pre-harvest (2004) and two years post-harvest (2005, 2006).

Species	Scientific Nomenclature	Species Code	Nesting Guild**	Habitat Guild***
Alder Flycatcher*	*Empidonax alnorum*	ALFL	S	E
American Goldfinch	*Carduelis tristis*	AMGO	T	E
American Redstart	*Setophaga ruticilla*	AMRE	T	E
American Robin	*Turdus migratorius*	AMRO	T	E
Baltimore Oriole	*Icterus galbula*	BAOR	T	E
Bay-breasted Warbler	*Dendroica castanea*	BBWA	T	I
Black-and-white Warbler	*Mniotilta varia*	BAWW	G	E
Black-capped Chickadee	*Poecile atricappilus*	BCCH	C	E
Brown-headed Cowbird	*Molothrus ater*	BHCO	n/a	E
Blue-headed Vireo	*Vireo solitarius*	BHVI	T	E
Blackburnian Warbler	*Dendroica fusca*	BLBW	T	I
Black-throated Green Warbler	*Dendroica virens*	BTNW	T	I
Blue Jay	*Cyanocitta cristata*	BLJA	T	E
Boreal Chickadee	*Poecile hudsonicus*	BOCH	C	E
Brown Creeper	*Certhia americana*	BRCR	C	E
Canada Warbler	*Wilsonia canadensis*	CAWA	G	I
Cedar Waxwing	*Bombycilla cedrorum*	CEDW	T	E
Chipping Sparrow	*Spizella passerina*	CHSP	T	E
Common Grackle	*Quiscalus quiscula*	COGR	T	E
Common Snipe*	*Gallinago delicata*	COSN	G	E
Common Yellowthroat*	*Geothlypis trichas*	COYE	G	E
Chestnut-sided Warbler	*Dendroica pensylvanica*	CSWA	S	E
Downy Woodpecker	*Picoides pubescens*	DOWO	C	E
Dark-eyed Junco	*Junco hyemalis*	DEJU	G	E
Eastern Kingbird*	*Tyrannus tyrannus*	EAKI	G	E
Gray Jay	*Perisoreus canadensis*	GRAJ	T	E
Hairy Woodpecker	*Picoides villosus*	HAWO	C	E
Hermit Thrush	*Catharus guttatus*	HETH	T	E
House Wren	*Troglodytes aedon*	HOWR	C	E

Appendix B1. (continued).

Species	Scientific Nomenclature	Species Code	Nesting Guild**	Habitat Guild***
Least Flycatcher	*Empidonax minimus*	LEFL	T	E
Lincoln's Sparrow*	*Melospiza lincolnii*	LISP	G	E
Magnolia Warbler	*Dendroica magnolia*	MAWA	G	E
Mourning Dove	*Zenaida macroura*	MODO	T	E
Mourning Warbler	*Oporornis philadelphia*	MOWA	G	E
Nashville Warbler	*Vermivora ruficapilla*	NAWA	G	E
Northern Flicker	*Colaptes auratus*	NOFL	C	E
Northern Waterthrush*	*Seiurus noveboracensis*	NOWA	G	E
Ovenbird	*Seiurus aurocapillus*	OVEN	G	I
Philadelphia Vireo	*Vireo philadelphicus*	PHVI	T	E
Pine Siskin	*Carduelis pinus*	PISI	T	E
Rose-breasted Grosbeak	*Pheucticus ludovicianus*	RBGR	T	E
Red-breasted Nuthatch	*Sitta canadensis*	RBNU	C	E
Red-eyed Vireo	*Vireo olivaceus*	REVI	T	E
Ruby-crowned Kinglet	*Regulus calendula*	RCKI	T	E
Ruffed Grouse	*Bonassa umbellus*	RUGR	G	E
Red-winged Blackbird*	*Agelaius phoeniceus*	RWBL	G	E
Sora*	*Porzana carolina*	SORA	G	E
Solitary Sandpiper*	*Tringa solitaria*	SOSA	G	E
Song Sparrow*	*Melospiza melodia*	SOSP	G	E
Spotted Sandpiper	*Actitis macularia*	SPSA	G	E
Swamp Sparrow*	*Melospiza georgiana*	SWSP	G	E
Swainson's Thrush	*Catharus ustulatus*	SWTH	T	E
Tennessee Warbler	*Vermivora peregrina*	TEWA	G	E
Tree Swallow*	*Tachycineta bicolor*	TRES	C	E
White-breasted Nuthatch	*Sitta sitta*	WBNU	C	E
Western Wood Peewee	*Contopus sordidulus*	WEWP	T	E
White-throated Sparrow	*Zonotrichia albicollis*	WTSP	G	E
Yellow-bellied Sapsucker	*Sphyrapicus varius*	YBSA	C	E
Yellow Warbler*	*Dendroica petechia*	YWAR	G	E
Yellow-rumped Warbler	*Dendroica coronata*	MYWA	S	E

*Riparian species
**Nesting guild: C- Cavity; G- Ground; S- Shrub; T- Tree canopy
***Habitat guild: E- Edge tolerant; I- Interior

CHAPTER 4. GENERAL CONCLUSIONS, MANAGEMENT IMPLICATIONS AND RECOMMENDATIONS FOR FUTURE RESEARCH

4.1 General Conclusions

Riparian areas are thought to have high conservation value due to their assumed function in reducing negative anthropogenic influences from uplands on aquatic systems and in providing habitat for a species-rich and productive biological community (Gregory et al. 1991, Harper and Macdonald 2001, Sarakinos et al. 2001, Hannon et al. 2002, Sabo et al. 2005). The most common approach to the management of boreal forest riparian areas is to retain treed buffer strips adjacent to them (see Lee et al. 2004). Many studies have investigated the suitability of different widths of treed buffers as habitat for upland-nesting birds. However, these studies have largely not focused on how species inhabiting the non-forested portion of the riparian area respond to disturbances (natural and anthropogenic) in the upland even though these species may be unduly affected by such disturbances.

This study was initiated to assess the response of birds using the riparian-upland interface in Boreal Plain riparian habitats of western Canada to two major disturbance types, fire and forestry. Specifically, the objectives of this thesis were to: 1) contrast bird communities inhabiting early post-fire and post-harvest boreal riparian areas, and; 2) determine which aspects of early post-disturbance riparian areas influence bird community composition. Some of the significant findings from my research along with management recommendations and suggestions for future research are outlined below.

One of the dominant tenets in sustainable forestry management of the North

American boreal is the Natural Disturbance Paradigm (Hunter 1993). This paradigm suggests that flora and fauna of the boreal forest have evolved adaptations to disturbance events, especially fire, and that the most effective means of managing for these species is by approximating such disturbance regimes through forestry practices. While many studies have investigated various aspects of this paradigm in upland boreal forests (see Hannon and Drapeau 2006, Schieck and Song 2006), a significant knowledge gap exists with respect to the effect of disturbances at the riparian-upland interface on bird communities. In Chapter 2, I contrasted bird communities inhabiting recently (1-4 years) burned and recently harvested boreal forests adjacent to riparian areas to test the efficacy of recent changes to forest management guidelines intended to approximate wildfire (10 m buffer with 25% retention in the next 30 m) along with sites that were harvested according to previous guidelines (30 m buffers). I also included wetlands with burned non-merchantable forests to determine the Natural Range of Variation of post-fire riparian bird communities.

Pre-disturbance digital forest inventories did not show large differences in vegetation composition between selected post-fire (merchantable sites) and post-harvest treatments indicating their appropriateness for contrast. This part of my thesis showed that fire is an important component of boreal riparian areas and that some riparian species (e.g., Alder Flycatcher *Empidonax alnorum*, Common Yellowthroat *Geothlypis trichas*, Le Conte's Sparrow *Ammodramus leconteii*) may prefer burned shoreline forest and riparian areas versus harvested shoreline forest habitats. Burned riparian areas showed a high range of variability in bird community composition that was not apparent in harvested shoreline forests. I found that new riparian management guidelines designed to approximate

patterns of wildfire in riparian areas did not produce overall (riparian and upland) bird communities similar to those in post-fire riparian habitats during the early post-disturbance phase. In fact, bird communities inhabiting larger buffers were more similar to bird communities in post-fire habitats presumably due to a greater range of habitat types available in larger buffers. I suggested that sites with smaller buffers may provide more suitable habitat for riparian species in later stages of succession (e.g., 5-10 year after harvest) due to greater similarities in vegetation structure (e.g., shrubs). Although multivariate analysis of riparian-associated species in the different treatments showed greater similarity than the overall bird community, significant differences were still found between treatments. The greatest dissimilarity occurred between burned and harvested bird communities. I concluded that because there is less physical change in riparian habitats than upland forest with both fire and forestry, riparian bird communities were not as influenced by these disturbances. Incorporating greater variability into riparian management than the current fixed-width buffer approach would likely be more beneficial to a greater suite of species by providing a greater range of available habitats.

The majority of riparian management guidelines followed in the commercial boreal forest mandate the retention of buffers 50 m wide or less on most water bodies (Lee et al. 2004). Since several recent studies have highlighted the ineffectiveness of these sizes of buffers in providing habitats for upland species (Lambert and Hannon 2000, Hannon et al. 2002). Therefore, I investigated riparian bird community change with smaller buffer widths in Chapter 3. The goal of this research was to determine if species specifically inhabiting non-forested riparian habitats were negatively influenced by these buffers relative to unharvested reference sites. Alternatively, I considered the possibility that

riparian bird species might benefit from disturbance of the upland forest buffer.

To accomplish this aspect of the thesis, I used a Before-and-After Controlled Impact (BACI) study design with three forest harvesting treatments and unharvested reference sites. The treatments were chosen to replicate a common fixed-width buffer management strategy (50 m buffers), guidelines recently implemented to approximate patters of natural disturbance (10 m buffer with 25% retention in the next 30 m) and a guideline used in Manitoba on ephemeral wetlands (0 m buffer). Sites were surveyed one year prior to harvest and two consecutive years after harvest. Individual species showed variable responses to the different management scenarios. Expectedly, interior forest-nesting species showed the greatest decline in abundance with decreasing forest retention. Riparian species including the Common Yellowthroat and Song Sparrow (*Melospiza melodia*) showed significant changes in abundance with year and harvest but showed no clear response to increasing amounts of harvest.

Analysis of bird communities in Chapter 3 using multiple response permutation procedures (MRPP) showed that the combined riparian and upland bird community changed with year post-disturbance and amount of harvest in riparian areas. However, bird communities showed a delayed response to harvest similar to other studies in the boreal forest (e.g., Schmiegelow et al. 1997, Warkentin et al. 2003). Differences were not found for the overall bird communities in reference sites over the three years indicating that the treatments were responsible for influencing bird community structure. Riparian communities associated with each harvest treatment and reference sites did not change with year or treatment indicating that riparian species were generally not influenced by level of harvest in the adjacent upland.

4.2 Management Recommendations

Based on my results, buffer management strategies currently being employed or considered in the boreal forest of western Canada may not have a great impact on species inhabiting non-forested riparian habitats (adjacent to shoreline forest) and some interior-forest nesting species. Increased abundance and frequency of occurrence of riparian species in post-fire vs. post harvest sites highlights that some species may prefer habitats generated from fire, especially early post-disturbance (i.e. 1-5 years). Consequences of eliminating or suppressing fires in boreal riparian areas are unknown but may reduce the diversity of bird communities typically associated with post-fire riparian habitats

Lack of a negative response by riparian species to various degrees of harvesting in shoreline forests allows forest managers to explore harvesting schemes that do not require buffer retention adjacent to all wetlands on the landscape. This approach would presumably permit greater flexibility in placement and configuration of residuals left in harvested landscapes rather than leaving older forests primarily in linear strips or small patches. However, I did not detect rare species sufficiently and these may be most at risk of population change due to forestry.

While I caution against the use of fixed-width buffers, I do not advocate the total removal of all buffers. Instead, a more flexible landscape-based approach that allows for the reduction or elimination of buffers in some areas and use of variable retention harvesting or selection harvesting so residuals can be retained in larger patches. This may aid in the conservation of upland-nesting species that require larger contiguous patches of older forest and potentially provide habitat for riparian species that utilize regenerating early seral stage habitats adjacent to riparian areas as a form of riparian habitat.

Placement of large forest patches adjacent to riparian areas should also be considered because some riparian species (e.g., Olive-sided Flycatcher *Contopus cooperi,* Northern Waterthrush *Seiurus noveboracensis*) may prefer such habitat and riparian areas are often more structurally heterogeneous and exhibit higher insect productivity than upland forest. However, further research in these areas is needed to understand bird community dynamics at the riparian interface with different shoreline forest harvesting strategies.

4.3 Recommendations for future research

There are several limitations to this study and it should be viewed very much as a preliminary evaluation of factors influencing the composition of riparian bird communities in the boreal. Many of the species (both riparian and upland) that were detected sufficiently for use in data analysis were habitat generalists. Rare species and those with larger territory sizes were not detected many times. Such species, including the Palm Warbler (*Dendroica palmarum*) and the 'threatened' Rusty Blackbird (*Euphagus carolina*; COSEWIC 2006), may be more vulnerable to landscape changes than many of the species found in this study (i.e., Common Yellowthroat, Song Sparrow). Focused studies on such species would help to direct forest management at the riparian and landscape scales to derive better conservation value.

While species' abundances are generally good proxies for habitat quality and breeding success (Bock and Jones 2004), this is not always the case (Van Horne 1983, Vickery 1992). Further studies should investigate the breeding success of selected riparian species to determine the effect of various management strategies relative to nest success in naturally disturbed and undisturbed riparian sites. For Chapter 2, I did not have

the opportunity to survey bird communities in sites with selectively harvested buffers or with buffers totally removed. This harvesting pattern would be more similar to post-fire riparian environments and may produce riparian bird communities more similar to those in burned merchantable sites.

Furthermore, additional research should experimentally burn riparian buffer strips and the adjacent non-forested riparian area to contrast experimentally burned post-fire bird communities with naturally burned sites since prescribed burns may not produce post-fire habitat conditions similar to natural fires (Smucker et al. 2005). I suspect that bird communities inhabiting prescribed burned riparian habitats would exhibit the greatest similarity to naturally burned riparian areas of any of the previously mentioned approaches to forest management because in the absence of fire, wetland riparian areas in harvested landscapes do not undergo significant successional change. Prescribed burning could then potentially be used as a management tool in Boreal Plain riparian areas and shoreline forests. Prescribed burns may also provide nutrient inputs into water bodies similar to natural fires (Nitschke 2005). Additionally, modeling the effects of current riparian management guidelines on old-growth forest, wood supply and amount of habitat would help direct future guidelines in forest management at a landscape scale. If riparian buffer strips represent the only old-growth on the landscape, then burning buffers will not be a viable management scenario.

Many aspects of the ecology of riparian birds are unknown including species-specific habitat requirements. Further study is needed to understand inter- and intra-annual variation in abundance of bird species and communities in riparian-upland ecotones in response to water-level changes (flooding and drawdown), temperature fluctuations and

changes in vegetation variability due to disturbance (e.g., herbivory by beaver *Castor canadensis*, other mammals and wind blowdown). We also have little idea of the contribution of aquatic insect emergence to riparian and associated upland bird communities and how variation in such insect abundance contributes to variation in species richness and abundance in those habitats. Stable isotopes possibly provide a convenient tool for measuring the relative contribution of aquatic insects to the upland avian food web (Hobson and Clark 1992).

Bird community composition of large vegetated boreal wetlands (e.g., marshes, fens, bogs) is virtually unknown. These areas may provide key breeding habitats for many passerine birds. Therefore, it is important to understand differences in productivity between these wetland types and those at the riparian-upland ecotone because differences in productivity could potentially direct shoreline forest harvesting. Additionally, wetlands are expected to be greatly impacted by climate change (Weltzin et al. 2000) and collecting baseline data on birds inhabiting various wetland types should be a priority to assess the impacts of climate change on riparian and wetland birds.

Riparian management guidelines for the Boreal Plain generally do not consider the impact of beaver harvest on wood supply and how they alter habitats adjacent to waterbodies. In my field research in the Duck Mountain study area, I anecdotally noted that beaver harvest changed the successional trajectory of shoreline forests dominated by hardwood forests to habitats dominated by shrubs (hazel *Corylus cornuta*, rose *Rosa spp*.). Further research should investigate the influence of beaver harvest on wood supply and habitat for species using the riparian ecotone. Coupled with greater incidence of tree blowdown, beavers may negatively influence the 'survival' of treed buffers maintained

after logging operations. Studies should investigate how individually marked trees in different sized buffers (versus reference sites) with varying forest composition survive over time by initiating tree "mark-recapture" experiments. This would help forest managers to understand the effectiveness of buffer-strip maintenance in providing habitat for wildlife through time. This is especially important if the majority of old-growth forest being maintained on the landscape is in buffer strips (Lee and Barker 2005).

The importance of detection probability estimation for abundance measures is well established in the current literature (e.g., White et al. 2005, Nichols et al. 2007). However, current methods for estimating detection probabilities are not valid for transects placed parallel to riparian habitats. Thus, for both chapters 2 and 3, I was not able to adjust transect species counts to account for changing detectabilities with varying habitat types, temporal effects, observers and distance. Since transects placed parallel to riparian habitats may be more effective for detecting riparian bird species than perpendicularly-placed transects or point counts, development of methods for estimating detection probabilities along transects placed parallel to riparian habitats will indeed allow researchers to derive more accurate estimates of abundance. Recent advances in the use of time of availability for detection of birds (Diefenbach et al. 2007) offer opportunities to use this method on transects. This technique is currently under development (Kardynal et al., unpublished data), but requires further testing and expert input to permit its use in future studies.

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